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Wilderness Conservation in the Anthropocene

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Abstract

Humanity is exerting unprecedented pressure on the natural environment, threatening tens of thousands of species with extinction, and degrading the condition of ecosystems worldwide. The extraordinary value of the last ecologically intact landscapes, free from industrial level degradation (which I call wilderness), is becoming increasingly recognised (**chapter 1**). Wilderness areas support many of the evolutionary and ecological processes that underpin all life on Earth, are critical for biodiversity conservation, and support the cultural integrity of many indigenous communities. The overarching aim of this thesis is to address key questions relevant to conserving wilderness areas and their values, with a focus on biodiversity conservation.

Despite the exceptional value of wilderness areas, information on their location, condition, and threat status globally has been limited. In **chapter 2**, I utilise a high-resolution (1km^2) global dataset of human pressure on the terrestrial environment for the years 1993-2009 (**appendix 1**) to develop the first temporally inter-comparable global maps of terrestrial wilderness areas. I define wilderness as places free from human pressures such as land clearing, dense human settlements, agriculture, and infrastructure developments, which significantly damage the environment. A spatial analysis of changes in wilderness extent between 1993 and 2009 showed catastrophic declines amounting to 3.3 million km^2, with the greatest losses occurring in the Amazon and Central Africa (**appendix 2**). Only 30 million km^2 of wilderness remains (23% of terrestrial areas).

Considering rates and extent of wilderness loss varies throughout the world, it is important to identify where this loss is impacting species. In **chapter 3,** I present a global analysis of cumulative human impacts on threatened species. I develop a novel spatial framework that jointly considers the co-occurrence of threats and the distribution of 5,457 vertebrates. I discover that human impacts extend across 84% of Earth's terrestrial surface, and identify 'hotspots' of impacted species richness. One quarter (n=1237) of species are impacted by threats across >90% of their distribution, and 395 species are impacted across their entire range. The methodology represents a conceptual advance for analysing threats to biodiversity, moving beyond analysing human pressures, which are agnostic to species type and their individual sensitivities to threats, to analysing realised impacts on individual species.

It is also important to analyse human impacts on the places set aside to protect biodiversity. In **chapter 4**, I present the first quantitative global assessment of the ecological condition of Natural World Heritage Sites (WHS), the world's flagship protected areas. I analyze changes in human pressure and forest loss within WHS, finding that many are more threatened than previously thought. Human pressures and forest loss occur in the vast majority of WHS, causing significant damage to the integrity of some sites. The results provide information to support the ongoing preservation of WHS to ensure they maintain their ecological integrity. The approach presents a transparent, defensible method for monitoring the ecological state of conservation areas.

Chapters 3, 4 and Appendix 2 highlighted that wilderness areas are declining and under-protected from the threats they face. They are also not recognised in any major international environmental agreements. Recognising the need for an international policy mechanism dedicated to wilderness conservation; in **chapter 5,** I argue that the World Heritage Convention could fill this gap. I assess wilderness coverage within WHS globally, and identify biogeographic regions without coverage ('gaps'). I then identify large, nationally designated protected areas with good wilderness coverage within gaps, which could potentially become new WHS. The results demonstrate that the Convention could make a substantial contribution to wilderness conservation by designating new wilderness WHS, and by protecting the wilderness condition of existing WHS.

The global analysis in **chapter 5** highlights the need for regional analyses that align with the scale of conservation action. In **chapter 6**, I present a regional case study where I analyze patterns of forest loss in an African wilderness area: Niassa National Reserve in Mozambique, where considerable effort and funding is going to conserving forest habitat and wildlife. I show that Niassa's forest loss is substantially lower than loss in the surrounding region, suggesting it is performing well at limiting forest loss relative to external pressure. The majority of Niassa's habitat is intact, and could support large mega-faunal assemblages. Its outstanding wilderness value could make it a good candidate for World Heritage Status.

This thesis highlights that our window of opportunity to safeguard wilderness areas and their values for people and nature is closing fast. I provide crucial information on the location, threat, and protection status of terrestrial wilderness areas globally and identify important places for species persistence, providing useful information to guide future conservation agendas at national and global scales.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, financial support and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my higher degree by research candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

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Publications included in this thesis

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Contributions by others to the thesis

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List of Abbreviations used in the thesis

CBD – Convention on Biological Diversity

IUCN – International Union for the Conservation of Nature

NNR – Niassa National Reserve

OUV – Outstanding Universal Value

PA – Protected Area

RMSE – Root Mean Squared Error

SDGs – United Nations Sustainable Development Goals

The Convention or WHC – The World Heritage Convention

UNESCO – United Nations Educational, Scientific and Cultural Organisation

WDPA – World Database on Protected Areas

WHS – Natural and Mixed World Heritage Sites

UNFCCC – United Nations Framework Convention on Climate Change

CHAPTER 1 Introduction

“If future generations are to remember us with gratitude rather than contempt, we must leave them something more than the miracles of technology. We must leave them a glimpse of the world as it was in the beginning”

Lyndon B. Johnson

Humans have significantly altered the majority of Earth’s terrestrial area (Venter et al. 2016c). Land uses such as pasture and cropping are now extensive (Ramankutty et al. 2008, Alkemade et al. 2013), industrial infrastructure, forestry, road networks and urban areas are proliferating (Seto et al. 2012, Ibisch et al. 2016), and activities such as the harvesting and hunting of wild plants and animals are highly unsustainable (Ripple et al. 2016). The increase in these “human pressures” has come at the expense of previously wild places, causing biodiversity and the condition of ecosystems to decline worldwide (Barnosky et al. 2012, Ceballos et al. 2017). However, human pressure varies across Earth’s surface, and some corners of the planet remain undisturbed by the impacts of modern society. These places can be considered wilderness, and contain the most intact ecosystems on Earth (Lesslie et al. 1988, Mackey et al. 1998, Watson et al. 2016c).

Wilderness does not mean “pristine” or “untouched” because nowhere meets this standard in an era of human forced climate change and global pollution (Barnes et al. 2009, Scheffers et al. 2016). Wilderness areas are also not exclusive of people, as human presence can and does occur within them, provided human impacts are minimal and do not compromise the ecological integrity of the area (Mackey and Claudie 2015, Larsen and Jaeger 2017). Ancient human societies occupied all continents except Antarctica and influenced the environment much more than previously thought. For example, prehistoric humans farming domesticated plants shaped tree communities in some of the remotest parts of the Amazon rainforest (Levis et al. 2017). However, the ecological footprint of these societies is challenging to measure and is likely minimal compared to the environmental impacts of modern industrial land-uses (Klein et al. 2009, Newbold et al. 2016).

There is room for debate as to what truly constitutes wilderness, and a variety of definitions have been developed by governments, non-government organisations and researchers, but there is no global consensus on the definition (Hawes et al. 2018). Most prominent definitions have several attributes in common; they identify wilderness by setting minimum thresholds of naturalness (the extent to which an area is unaffected by human pressures or to which it retains its ecological integrity), remoteness (from infrastructure and landscape disturbance), and size (e.g. a minimum contiguous area) (Lesslie et al. 1988, Mackey et al. 1998, Hawes et al. 2018). For the purpose of this thesis, I follow these definitions and consider wilderness areas as “large, ecologically intact landscapes”. In order to map wilderness areas in **chapter 2**, I set minimum thresholds on the size, and naturalness of wilderness areas, providing precise definitions.

There have also been deliberations about whether the term wilderness is still relevant given humanity’s ubiquitous fingerprint (Cronon 1996). However, regardless of how wilderness is defined, it is clear that large intact ecosystems hold an exceptional range of environmental and cultural values that are being lost in more human modified, fragmented, and ecologically degraded landscapes (Freudenberger et al. 2012, Lovejoy 2017, Watson et al. 2018a). I now describe these values, and their important contribution to human well-being and the health of the planet, demonstrating that protecting wilderness is critical for averting the biodiversity crisis, addressing anthropogenic climate change, ensuring ecosystem service provisioning, and maintaining the world’s human cultural diversity.

The Exceptional Values of Wilderness

Biodiversity Conservation

Humans are currently driving a biodiversity extinction crisis. Species populations are declining, their ranges contracting, and extinction rates are 1000 times higher than background levels (Pimm et al. 2014, Ceballos et al. 2015, Ceballos et al. 2017). When it comes to averting this crisis, protecting wilderness areas is critical because they still support high levels of species richness and endemism, and contain large populations of common but declining species (Mittermeier et al. 2003, D’agata et al. 2016, Watson et al. 2016c). Wilderness areas are the only places that still contain species assemblages at near natural levels of abundance, especially for large mega-fauna (Mittermeier et al. 2003, Ripple et al. 2015), and wide ranging and migratory species (Klein et al. 2009, Bauer and Hoye 2014, Lamb et al. 2018, Photopoulou 2018). They also support the ecological processes that sustain biodiversity over evolutionary timescales such as trophic relations at regional scales and animal dispersal and migration (Soule et al. 2004b).

Ensuring large wilderness areas are protected is essential because smaller patches of habitat tend to lose many of their species over time (MacArthur and Wilson 1967, Soule et al. 2004b). This is especially true for top predators, whose populations frequently collapse in isolated habitat patches that are too small or suffer hunting from humans along the periphery. As such, wilderness areas are key refuges for species that are sensitive to exploitation by, or conflict with humans, which includes many charismatic carnivore species (Gibson et al. 2011, Ripple et al. 2014). As disturbance sensitive species disappear from human dominated landscapes, wilderness areas are rapidly becoming their last strongholds. These wilderness refugia serve as important reservoirs of genetic information, and sources of populations and propagules for restoration and rewilding efforts (Ceaușu et al. 2015). They are also places where nature can be studied free from human disturbance, serving as important baseline references of ecological and evolutionary function.

It is important to note that wilderness areas are not safe from all possible threats to biodiversity. For example, invasive species can spread into wilderness areas jeopardising biodiversity in these places. Wilderness areas will not necessarily protect species from climate change, and conditions in many of the most remote intact areas globally are changing rapidly. However, wilderness areas do confer many benefits to species in the face of climate change that other more degraded areas do not. Wilderness refuges are especially important given rapid climate change, because they allow species to adapt by housing large populations (allowing for local genetic adaptation), and enabling species to disperse to favourable climates without having to cross human barriers (Scheffers et al. 2016, Tucker et al. 2018). There is also evidence that the impacts of climate change on ecological communities are felt more severely in degraded and fragmented landscapes than intact ones, and this holds for many ecosystems, including rapidly changing ones such as the arctic (Hansen et al. 2001, Djoghlaf 2008, Mantyka-Pringle et al. 2012, Watson et al. 2018a). As such, proactively securing the remaining wilderness is increasingly seen as fundamental to global efforts to avert the biodiversity extinction crisis (Watson and Venter 2017).

Ecosystem services

Unimpeded by human activity, wilderness areas continue to support the natural evolutionary and ecological processes that underpin life on Earth, providing a suite of high value ecosystem services (Klein et al. 2009, Watson et al. 2009, IPCC 2014). For example, wilderness areas regulate climate regimes and hydrological cycles at multiple scales (Salati et al. 1979, Furniss et al. 2010). This includes generating rainfall, for example air that passes over ecologically intact tropical forests produces twice as much rain as degraded forests or converted land (Sheil and Murdiyarso 2009). Evapotranspiration also has a cooling effect that buffers the temperatures of local climates during heatwaves (Deo et al. 2009, Ahlström et al. 2015). The quantity of ecosystem services provided also increases with the size of a wilderness area (Wright et al. 1999, Chagnon and Bras 2005), and in many cases is a direct result of that size because it allows wilderness areas to act as complete self-organising systems (Sanderson et al. 2002). This has major implications for conserving wilderness areas because damage in one part of the system can affect the functioning of the entire system (Laurance 2005, Peres 2005). For example, it is estimated that the Amazon requires at least 75-80% of its forest cover to retain its hydrological cycle (Sampaio et al. 2007, Lovejoy and Nobre 2018), and therefore must be conserved almost in its entirety (Laurance 2005).

Proactively protecting carbon rich wilderness also makes a significant contribution to stabilising atmospheric CO_2 levels, and achieving global climate mitigation goals. This is because ecologically intact ecosystems have a greater capacity to sequester and store carbon than degraded ecosystems (Mackey et al. 2013, Avitabile et al. 2016). For example, one third of the total global stock of forest biomass carbon is stored in the boreal forest biome, the most intact ecosystem on the planet (Pan et al. 2011), and the Amazon region stores nearly 38% of the above ground carbon stored in woody vegetation in tropical America, Africa and Asia combined (Walker et al. 2014). Beyond forests, intact coastal systems including mangroves, salt marshes, and seagrasses have an exceptional capacity to sequester and store carbon (Fourqurean et al. 2012). However, when degraded, they quickly change from carbon sinks to major carbon sources (Howard et al. 2017). The same applies for other intact ecosystems, for example, slight degradation of intact forests can cause substantial carbon release (Houghton 2012, Chaplin-Kramer et al. 2015, Watson et al. 2018a).

Wilderness areas have an important role to play in the fight against anthropogenic climate change beyond storing and sequestering carbon. In many situations they provide humanity with a direct defence against extreme climatic events such as floods, sea level rise and cyclones (The World Bank 2009). For example, intact mangroves, grasslands, coral reefs, wetlands and forests are humanity's best protection against floods and storms (Bradshaw et al. 2007b, Ferrario et al. 2014), with highly intact ecosystems providing stronger protection than degraded ones (Alila et al. 2009, Brookhuis and Hein 2016). The intact mangroves of the Tamil Nadu coast in India, for instance, significantly reduced damage to human structures following the tragic Indian Ocean Tsunami of December 26th 2004 (Alongi 2008). Securing intact ecosystems is also recognised as humanity's most cost-effective defence against climate change (The World Bank 2009, Martin and Watson 2016), and is often orders of magnitude cheaper than engineered solutions such as building sea walls instead of protecting intact reefs, salt marshes and mangroves (Shepard et al. 2011, Jones et al. 2012). Recent analyses estimated that coral reefs save Indonesia, the Philippines, Malaysia, Mexico and Cuba over 400 million USD annually through flood avoidance (Beck et al. 2018), and that wetlands saved the United States over 625 million USD in flood damages following hurricane Sandy in 2012 (Narayan et al. 2017). Perversely, ecosystems such as coral reefs are often destroyed or degraded in the process of building sea walls and protective infrastructure (Grantham et al. 2011, Maxwell et al. 2015b).

Human Cultural Diversity

Wilderness areas are home to the most politically and economically marginalized indigenous communities on Earth (Gorenflo et al. 2012, Schwartzman et al. 2013). These peoples number in the hundreds of millions and are reliant on the ecosystem services provided by intact marine and terrestrial ecosystems for essential resources such as food, water, and fibre (MEA 2005). Securing wilderness is central to reducing poverty and marginalisation of these people's, and for achieving many United Nation's Sustainable Development Goals (SDG's) such as human well-being, clean water, reduced inequalities, peace and justice, as well as the biodiversity and climate goals. Many indigenous peoples have inhabited wilderness areas for millennia, developing strong bio-cultural connections to the land (Larsen and Jaeger 2017). In these cases, securing wilderness is critical for securing their long-term cultural integrity.

There have previously been conflicts between the people living in wilderness areas and conservationists, with wilderness debates historically framed as mutually exclusive choices between conservation and development. However, wilderness conservation thinking has evolved over the last few decades, and it is increasingly recognized that there are major common interests between conservationists and local communities in light of growing human pressure on the natural environment (Venter et al. 2016c, Larsen and Jaeger 2017).

The sustainable livelihoods and cultural integrity of many indigenous communities are often threatened by the same industrialized development pressures that threaten biodiversity and wilderness areas (Boff 2002, Sutherland 2003). For example, human cultural and language diversity, which co-occurs with biodiversity, is declining outside of wilderness areas (Gorenflo et al. 2012, Amano et al. 2014). There are also cases of violent conflict between economic development and indigenous peoples (Simmons 2002, Finer et al. 2008), including the mass-murder of uncontacted Amazonian tribes by gold miners (Darlington 2017), and inter-tribal warfare spurred on by oil exploration in Ecuador (The Economist 2013). Protected areas, indigenous protected areas, and other land-use designations preventing industrial development can act as important buffers protecting indigenous people and ensuring they have the wild spaces required to maintain their cultures. As such, there are opportunities to integrate both nature and culture in the conservation of large wilderness areas.

The State of Wilderness Conservation

Even though most definitions of wilderness lack precision, the broad concept of wilderness as still proved valuable, motivating conservation efforts for centuries. For example, wilderness inspired the writings of Audubon, Muir, Leopold and others, whose works form the moral foundations of our field. Leopold in particular recognised that landscapes are systems, which if functioning properly, support the processes that generate soil, water, and life. Leopold compellingly articulated this ethical philosophy in his 1949 classic *A Sand County Almanac*, popularising the idea that wilderness areas have value if preserved in their wild state, and inspiring generations of conservationists. Wilderness protection was also the inspiration for Yellowstone, the world’s first national park, designated in 1872 in the United States. The national park model spread worldwide and still forms the backbone of conservation efforts today.

Despite the historical importance of wilderness, and the emerging scientific consensus showing that intact ecosystems have exceptional ecological values compared to more degraded ecosystems, wilderness areas been overlooked as conservation priorities in the last few decades (Myers et al. 2000, Brooks et al. 2006). This is partly because the concept of wilderness has come under heavy scrutiny, being described as a human construct, and criticised for promoting a dualistic vision which separates people from nature (Cronon 1996, Sarkar 1999). Wilderness conservationists have also been criticised for protecting wilderness from the people who live there, or at their expense, although modern wilderness conservation is now moving beyond this (Mackey and Claudie 2015, Larsen and Jaeger 2017). As a result of this criticism, the term wilderness has become controversial and even unmentionable in some circles (Hawes et al. 2018). However, it is important to note that these are not criticisms of wild places and their values, but rather how the term wilderness is defined and used, and how wilderness conservation is carried out (Cronon 1996).

There is often an implicit assumption amongst conservationists that wilderness areas are free from human impact due to their size, remoteness, and historically limited human presence. As a result, there is a notion that they do not require conservation action, and that conservation resources would be better spent on places that contain high concentrations of threatened species and are thought to be under more imminent threat (Margules and Pressey 2000, Ricketts et al. 2005). Some conservationists have argued that such reactive approaches to conservation should take precedence over proactive efforts to conserve wilderness (Ricketts et al. 2005, Kareiva and Marvier 2012, Pressey et al. 2017). However, it is increasingly acknowledged that this is a false dichotomy and conservation requires a balance of both reactive and proactive approaches (Brooks et al. 2006, Cardador et al. 2015, Watson and Venter 2017). Furthermore, when this thesis was conceptualised in 2015, there were no global analyses of changes in the extent of wilderness areas over time, making it challenging to ascertain how threatened they were.

Information on the condition, threat and protection status of wilderness areas globally is limited because they are not monitored in any regular fashion (Lovejoy 2017), and the best available maps of global wilderness extent date back to the early 2000’s (Hannah et al. 1994, Sanderson et al. 2002, Mittermeier et al. 2003). Although these maps proved useful for numerous ecological and conservation analyses (Di Marco and Santini 2015, Inostroza et al. 2016, Kormos et al. 2016, Payne and Bro-Jørgensen 2016), they now provide a temporally static and much outdated view of wilderness extent (Watson et al. 2009, Laurance et al. 2012, Laurance et al. 2014).

Despite a lack of up-to-date information, concerns for the safety of wilderness areas take validity from the widespread pressure humanity is exerting on the natural environment (Steffen et al. 2015a, Venter et al. 2016c). Wilderness values can be lost long before habitat has been completely cleared, making wilderness areas particularly vulnerable to pressures such as roads that can have extensive indirect or ‘offsite’ impacts (Raiter et al. 2014, Raiter et al. 2018). These can include the isolation and fragmentation of species populations, increased access for hunting and logging, and increased risk of anthropogenic fire ignition (Mackey et al. 1998, Laurance et al. 2017). In light of growing human pressure globally, there have been recent calls within the conservation community for updated maps of wilderness areas, which would make global assessments of their threat and protection status possible (Bertzky et al. 2013, Kormos et al. 2016).

Recognising that the lack of up-to-date information on wilderness extent constituted a major impediment to wilderness conservation, in **chapter 2**, I developed new global maps of terrestrial wilderness areas for the years 1993 and 2009. This data enabled an analysis of changes in wilderness extent during this time period, which highlighted the catastrophic loss of 3.3 million km^2 of wilderness. This is an area greater in size than India, which amounts to a 10% decline in total global wilderness extent over 16 years (**appendix 2**). To give this context, wilderness loss is occurring four times faster than forest loss globally (Keenan et al. 2015), making wilderness areas some of the most threatened ecosystems globally. Only 30 million km^2 of Earth's terrestrial wilderness remains (23%) and it is rapidly diminishing. I mention the data generated in **chapter 2** and the findings of **appendix 2** at this point because they highlight important questions for wilderness and biodiversity conservation, and justify much of the work in **chapters 3-6** of this thesis. **Appendix 2** challenges the widely held perception that wilderness areas are relatively safe from significant human impacts. Many of the greatest wilderness declines occurred in the Amazon and Central Africa, which are some of the most biodiverse regions on Earth. **Appendix 2** also highlighted that efforts to conserve wilderness within protected areas between 1993 and 2009 lagged well behind the rate of loss, which was double the rate of protection globally.

It is also evident that wilderness conservation has received insufficient attention in international environmental agreements and policy deliberations to date (Watson et al. 2016c). Global efforts to achieve ambitious sustainable development and environmental targets have almost completely overlooked the contributions of wilderness areas and their outstanding values. At the international level, there is no formal recognition of the importance of wilderness areas, and no explicit targets for wilderness protection in many of the most powerful multilateral environmental agreements. This includes in the Convention on Biological Diversity (CBD 2011), a platform which attempts to coordinate international action to halt or reverse biodiversity loss (CBD 2011), and the World Heritage Convention, which aims to protect the world's most valuable natural and cultural sites (UNESCO 1972). It is also surprising that the United Nations Framework on Climate Change (UNFCCC) Paris

Agreement ignores the important role that wilderness areas play in the fight against climate change.

The little recognition that wilderness areas have received in policy includes the United States Wilderness Act of 1964 which established standards for protection of wilderness on federal land. A small number of countries including Canada, Australia, Finland and South Africa have followed suit, implementing similar legislation to protect wilderness (Mittermeier et al. 2003). The International Union for the Conservation of Nature (IUCN) recognises specific protected areas worldwide that contain wilderness, and includes criteria on size and intactness (Category C) in its recently published Global Standards for identifying Key Biodiversity Areas (IUCN 2016), but does not specifically mention the word wilderness.

The lack of strong international policy recognition for wilderness areas has significant ramifications for national environmental strategies. The tendency for national biodiversity conservation plans is to focus on remnant habitats and endangered populations of species living in degraded, fragmented and altered ecosystems, with very few nations clearly articulating conservation goals for wilderness areas (Klein et al. 2009, Watson et al. 2009, Ceauşu et al. 2015). Some wilderness areas are protected under national legislation such as the 1964 United States Wilderness act, which formally defines and protects wilderness on federal land. However, in most countries, wilderness areas are not formally defined, mapped or protected, and there is nothing to stop them being exploited by national and local governments, private business, and civil society.

The implications of the lack of policy recognition for wilderness areas has been noticed, leading to recent and timely calls for international environmental agreements to take a more central role in wilderness conservation (Martin and Watson 2016, Watson et al. 2016c, Watson and Venter 2017, Barnes et al. 2018, Maron et al. 2018). In particular, there have been calls for the World Heritage Convention, one of the most powerful conservation instruments, to become more engaged in wilderness conservation (Kormos et al. 2016). However, the exact roles that certain international agreements and/or conventions could play remains ambiguous, and efforts to assess their contributions, for example by examining the World Heritage Convention's coverage of wilderness areas, have been hindered until now by the absence of up-to-date spatial data on wilderness extent (Bertzky et al. 2013, Kormos et al. 2016)

Thesis Structure

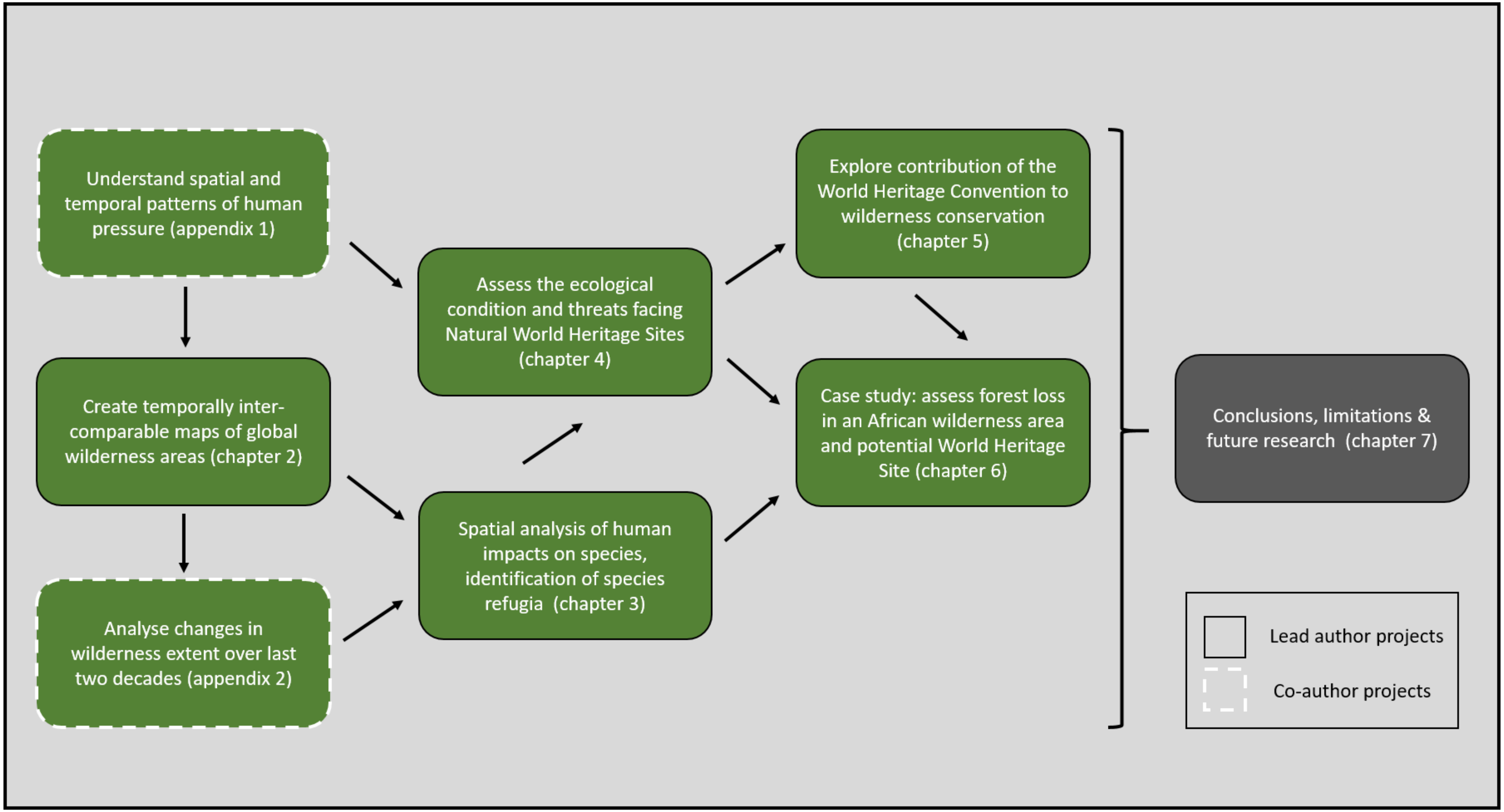
There is an urgent need for up-to-date global maps of wilderness areas to support conservation efforts. One method of identifying wilderness areas is to understand the spatial patterns of the human pressures that threaten them, then use the logic that places free from human pressure constitute wilderness (Sanderson et al. 2002). In **appendix 1**, I describe a new temporally inter-comparable spatial dataset of cumulative human pressure on the global terrestrial environment for the years 1993 and 2009. I was part of the international team that developed these datasets, (known as the updated "Human Footprint"), which underpin the work in **chapters 2-5** of this thesis. The Human Footprint includes data at a 1km^2 resolution globally on eight human pressures including: built environments, crop lands, pasture lands, population density, night lights, railways, major roadways and navigable waterways, making it the most comprehensive cumulative pressure or 'threat' map available (McGowan 2016). The data also exhibit an excellent degree of accuracy, which was determined via a rigorous validation process. High resolution satellite imagery was used to interpret human pressures in >3000 randomly selected plots across Earth's surface. These were then statistically compared with the corresponding human footprint data to test for agreement.

In **chapter 2**, I develop the first temporally inter-comparable global maps of terrestrial wilderness areas. To do this, I use the Human Footprint data (**appendix 1**) and identify wilderness as all pressure free lands with a contiguous area > 10,000 km^2 for the years 1993 and 2009. These places represent the most intact ecosystems globally, and are likely to be operating with natural levels of species abundance, community structure and disturbance regimes. Recognising that human pressure differs substantially in intensity across Earth's surface, and that wilderness values can exist in regions with some level of human pressure. I also create a regionally representative map of wilderness following the well-established "Last of the Wild" methodology (Sanderson et al. 2002). This method involves identifying the 10% area with the lowest human pressure in 60 of Earth's biogeographic realms, then identifying the ten largest contiguous areas, along with all contiguous areas >10,000km^2. I discuss the benefits of these techniques for mapping wilderness areas relative to other recent efforts to identify ecologically intact and valuable regions (Potapov et al. 2017), and discuss the many potential applications of the data. The wilderness data underwent a similar visual validation to the human footprint data and also exhibit an excellent degree of accuracy. I limited this analysis to the terrestrial realm because although the concept of wilderness is still relevant in marine systems, it would require another entire literature review on the value of marine wilderness areas, and would require utilizing several other big datasets on human pressure on marine environments. I therefore believe mapping marine wilderness warrants its own stand-alone study is beyond the scope of this thesis.

As mentioned previously, in **appendix 2** I describe an analysis of changes in wilderness extent between the years 1993 and 2009, highlighting that wilderness is rapidly declining, with many of the greatest losses occurring in biodiverse regions such as the Amazon. Considering this variation in wilderness loss, and the multiple threats driving it, it is important to identify where species are most impacted, and where key refuges are located. To do this, in **chapter 3**, I present the first global spatial analysis of cumulative human impacts on threatened species. It is important in this work to recognise the difference between pressures and impacts. Pressures are defined as human actions with the potential to induce environmental change, whilst impacts are the negative effects on the environment, which are caused by human pressures (Martins et al. 2012). Here, I develop a novel spatial framework that jointly considers the co-occurrence of threatening processes (data in **appendix 1**) and the known vulnerabilities of individual species with the geographical distributions of 5,457 threatened terrestrial vertebrates. The framework represents a conceptual and methodological advance for how we analyse threats to biodiversity, by moving beyond just analysing human pressures, which are agnostic to species type and sensitivity to threats, to analysing realised impacts on individual species (Martins et al. 2012, Halpern and Fujita 2013). I identify global hotspots of impacted and unimpacted species richness, and using similar logic to the wilderness mapping in **chapter 2**, I assume that places free of impact constitute species refugia. This is important information for global conservation efforts aimed at proactively securing intact landscapes that are critical for threatened species conservation, and for reactive conservation efforts aimed at abating threats.

Having analysed human impacts on wilderness areas and threatened species, I now analyse impacts on some of the places set aside to protect them. In **chapter 4**, I present the first globally standardised quantitative assessment of the ecological condition of Natural World Heritage Sites (WHS), the world's flagship protected areas. To do this, I analyse changes in the spatial and temporal patterns of human pressure (data in **appendix 1**), and forest loss (Global Forest Watch data), across the global World Heritage estate. I identify WHS that have suffered the greatest forest loss and increases in human pressure, which require immediate management intervention, as well as WHS that are performing well at limiting these negative changes. The results provide information to support the ongoing preservation of WHS to ensure they maintain their Outstanding Universal Values, which are often directly linked to their ecological integrity and wilderness condition. The approach also presents a transparent, defensible method for monitoring the ecological state of conservation areas. The work was conducted in collaboration with members of the International Union for the Conservation of Nature (IUCN), which is the official scientific advisory body to UNESCO on matters of Natural World Heritage, to ensure the work was relevant and informative for World Heritage related policy.

Figure 1.1 Thesis structure



Recognising the need for global environmental agreements to play a more substantial role in wilderness conservation, in **chapter 5,** I explore the potential contribution of the World Heritage Convention to wilderness conservation. To do this, I utilise the 'Last of the Wild' maps presented in **chapter 2** to assess current wilderness coverage within WHS globally, and to identify biogeographic regions where WHS have no wilderness coverage ('coverage gaps'). I then identify large, nationally designated protected areas with good wilderness coverage within coverage gaps, which could potentially be designated as new wilderness WHS if they meet the other conditions of the Convention. I discuss the conservation tools available under the Convention, and how these can be leveraged in their current form to help protect wilderness areas. This work was also conducted in collaboration with members of the IUCN, supporting the development of the IUCN report on "World Heritage, Wilderness, and Large Landscapes and Seascapes", which is an official UNESCO guidance document on wilderness conservation.

The global analyses in **chapters 4** and **5** highlighted the need for case studies that better align with the scale of conservation action. Therefore, in **chapter 6**, I examine patterns of forest loss in one of Africa's wilderness areas, Niassa National Reserve in Northern Mozambique, where substantial funding and effort is going into protecting its wilderness and wildlife. Niassa National Reserve is a protected area that has excellent wilderness coverage and falls within one of the gaps in the World Heritage Convention's coverage of wilderness identified in **chapter 5**. Therefore, Niassa is a case study of a protected area that could potentially be granted World Heritage Status based on its wilderness attributes, which would help improve the WHC's wilderness coverage. This work supported the development of a management plan for Niassa National Reserve, and was carried out in collaboration with the Wildlife Conservation Society, who co-manage Niassa with the Mozambican government.

Finally, in **chapter 7**, I discuss the major conclusions from each chapter and their significance for wilderness and biodiversity conservation. In this synthesis, I describe some of the main messages to emerge from this work, some of the limitations of the research, and present suggestions for future research. In **chapters 2-6,** I have retained the text consistent with their published form. Therefore, I use the plural "we" instead of the more commonly used "I", since each chapter is a collaborative multi-authored paper. Each chapter is written in the style of the journal in which it is published so there are slight differences in formatting between chapters. Finally, there is some repetition among chapters in their introductions, which is necessary for them to stand alone as papers.

CHAPTER 2 Temporally inter-comparable maps of terrestrial wilderness and the Last of the Wild

Allan, J.R. Venter, O. Watson, J.E.M.

Abstract

Wilderness areas, defined as areas free of industrial scale activities and other human pressures which result in significant biophysical disturbance, are important for biodiversity conservation and sustaining the key ecological processes underpinning planetary life-support systems. Despite their importance, wilderness areas are being rapidly eroded in extent and fragmented. Here we present the most up-to-date temporally inter-comparable maps of global terrestrial wilderness areas, which are essential for monitoring changes in their extent, and for proactively planning conservation interventions to ensure their preservation. Using maps of human pressure on the natural environment for 1993 and 2009, we identified wilderness as all 'pressure free' lands with a contiguous area >10,000km^2. These places are likely operating in a natural state and represent the most intact habitats globally. We then created a regionally representative map of wilderness following the well-established "Last of the Wild" methodology; which identifies the 10% area with the lowest human pressure within each of Earth's 60 biogeographic realms, and identifies the ten largest contiguous areas, along with all contiguous areas >10,000km^2.

Background & Summary

Wilderness areas are ecologically intact landscapes free of human pressures which cause significant biophysical disturbance of the natural environment (Lesslie et al. 1988, Mackey et al. 1998). This includes industrial activities such as land-clearing, dense human settlements, agriculture, industry, and infrastructure development (Mittermeier et al. 2003, Watson et al. 2016c). Importantly, this definition does not exclude indigenous peoples and communities, who have been part of wilderness areas for millennia through deep bio-cultural connections to the land (Gorenflo et al. 2012, Mackey and Claudie 2015).

Natural ecological and evolutionary processes continue largely unimpeded in wilderness areas, providing a suite of high-value ecosystem services (Watson et al. 2009, Martin and Watson 2016). These include regulation of hydrological cycles at multiple scales (Salati et al. 1979, Furniss et al. 2010, Martin and Watson 2016), and significant organic carbon stocks (Mackey et al. 2013, Lovejoy 2017). Wilderness areas are also critically important for in situ biodiversity conservation, supporting the last intact mega-faunal assemblages (Mittermeier et al. 2003, Ripple et al. 2015), wide ranging and migratory species (Klein et al. 2009, Bauer and Hoye 2014), and species sensitive to exploitation by or conflicts with humans (Ripple et al. 2014). Wilderness areas are also the last remaining places on Earth where scientists can study biodiversity and natural processes free from the influence of modern society.

Maps of terrestrial wilderness areas have previously been developed by mapping the extent of a number of human pressures on the environment at both global and regional scales (Sanderson et al. 2002, Mittermeier et al. 2003, Inostroza et al. 2016), using the logic that the areas free of human pressure constitute ‘wilderness’. These maps have proved useful for numerous ecological and conservation analyses (Di Marco and Santini 2015, Inostroza et al. 2016, Kormos et al. 2016, Payne and Bro-Jørgensen 2016). However, these maps provide a temporally static and now much outdated view of wilderness extent (Watson et al. 2009, Laurance et al. 2012, Laurance et al. 2014), and there have been recent calls for a more updated product (Kormos et al. 2016).

Here we present two new data-sets of spatially and temporally intercomparable maps of global terrestrial wilderness areas for the years 1993 and 2009. We used the methodological framework outlined in the original ‘Last of the Wild’ work (Sanderson et al. 2002) but utilized the recently updated ‘Human Footprint’ maps (Venter et al. 2016a). These are the most up-to-date and highest resolution globally standardized maps of cumulative human pressure on the terrestrial environment (Venter et al. 2016c). The Human Footprint is the only pressure map to have had its data validated (Venter et al. 2016a), and is widely regarded as the best available product of its kind (McGowan 2016).

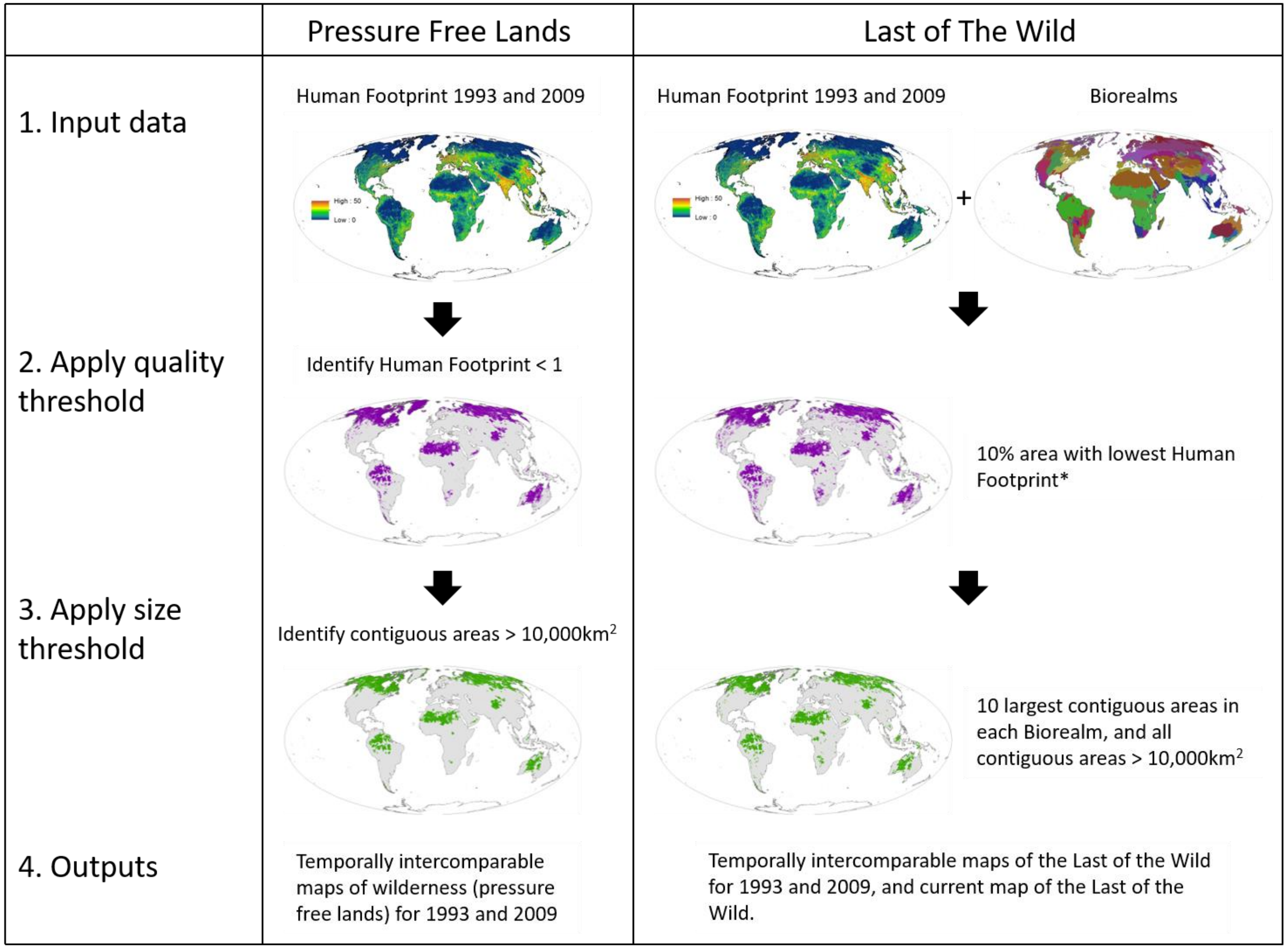
Our maps of wilderness areas have already been used to highlight catastrophic declines in wilderness extent over the last two decades, and show that conservation efforts has been greatly outpaced by these losses (Watson et al. 2016c). This has raised the profile of wilderness conservation globally (Belote et al. 2017, Lovejoy 2017), and it seems that international targets for wilderness conservation may be developed shortly (Kormos et al. 2016, Lovejoy 2017). We anticipate that our maps will be important tools in the process of developing such targets, and for the conservation planning and decision making necessary to ensure representative protection of wilderness areas globally (Mittermeier et al. 2003, Watson et al. 2016a, Allan et al. 2017b).

Methods

The Human Footprint

To map the global extent of wilderness we utilised the recently updated Human Footprint maps for 1993 and 2009 (Venter et al. 2016a, c) (Fig. 2.1). These are globally-standardised maps of cumulative human pressures on the terrestrial environment. At a 1km^2, they are the finest resolution cumulative threat maps available, as well as the most comprehensive, including data on eight human pressures globally: built environments; crop lands; pasture lands; population density; night-time lights; railways; major roadways; and navigable waterways. Following the original Human Footprint methodology (Sanderson et al. 2002), individual pressures were placed within a 0 – 10 scale based on their estimated contribution to human pressure, and summed giving a cumulative score ranging from 0 – 50 for each pixel (some pressures are mutually exclusive, whilst others can co-occur). We converted the Human Footprint datasets from a continuous to an integer 0 – 50 scale by truncating. The integer Human Footprint datasets were used for all the analyses described in the paper. The following sections describe in detail how these datasets were handled to map pressure free lands and the Last of the Wild.

Figure 2.1 Workflow of our approach to mapping pressure free lands and the Last of the Wild. * For temporally inter-comparable maps of the Last of the Wild the 10% threshold is based on the 1993 Human Footprint for both the 1993 and 2009 maps. For the current Last of the Wild the 10% threshold is based on the 2009 Human Footprint. See methods for more detail.



Comparable maps of pressure free lands for 1993 and 2009

We created two global maps of wilderness in 1993 and 2009 by identifying all areas which are free of human pressure (Human Footprint = 0), and have a contiguous area >10,000km^2. This size threshold has been used by others to identify wilderness areas (Mittermeier et al. 2003, Watson et al. 2009, Kormos et al. 2016), and is consistent with the parameter values for identifying intact ecological communities in the International Union for Nature Conservation (IUCN) standards for identifying Key Biodiversity Areas (IUCN 2016). Large wilderness areas separated by small areas of Human Footprint greater than '0' were treated as two discreet wilderness blocks. Given the difficulty in restoring wilderness condition, locations which had a Human Footprint score > 0 in 1993 but = 0 in 2009 were excluded, as was Antarctica for its lack of suitable data.

Temporally inter-comparable maps of the "Last of the Wild" for 1993 and 2009

We also created global maps of the "Last of the Wild" for 1993 and 2009 following the methodology developed by Sanderson et al (Sanderson et al. 2002). First, we created a layer of biogeographic realms (hereafter simply 'biorealms') as a biogeographic framework for our analysis, based on the widely used Terrestrial Ecoregions of the World (Olson et al. 2001). The biorealms represent combinations of the world's 14 vegetated biomes and seven biogeographic realms (for example boreal forests exist in both the Palearctic and Nearctic realms). Following established practice we excluded Antarctica and other rock and ice ecoregions (Juffe-Bignoli et al. 2014, Venter et al. 2014a). Our resulting map contained 60 out of a possible 67 biorealms because some sub-Antarctic and Pacific islands fall beyond the extent of the Human Footprint data.

We calculated biorealm specific thresholds on the 1993 Human Footprint scale which ensured that at least 10% of each biorealm's land area with the lowest Human Footprint in 1993 was captured. We then selected the ten largest contiguous blocks in each biorealm and all contiguous areas >10,000km^2 to create the Last of the Wild dataset for 1993. The same biorealm specific thresholds identified for the 1993 map for the 10% area with the lowest Human Footprint score for 1993 were also used to map the 2009 Last of the Wild so that it is possible to directly compare changes in wilderness extent across the two time periods. Finally, we created a map of the Last of the Wild for 2009 where we calculated the biorealm specific thresholds on the 2009 Human Footprint scale which ensured that at least 10% of a biorealms land area with the lowest Human Footprint in 2009 was captured (for the previous maps we used the 1993 threshold to ensure maps from the two time periods are comparable). This map is not comparable with the 1993 map, but is important since it shows the current best quality habitat left in all the biorealms.

Data Records

The 1km^2 resolution, temporally inter-comparable maps of pressure free lands and the 1993 and 2009 Last of the Wild maps are stored in the Dryad Digital Repository where they can be accessed freely. The Dryad files can be downloaded as a single 7-zip file archive which contains an individual shapefile (.shp) for each of the five maps and excel databases containing the validation data. The Human Footprint dataset which underpins this work is also freely available on Dryad and contains the entire dataset for the visual validation as well as the Human Footprint maps.

Technical Validation

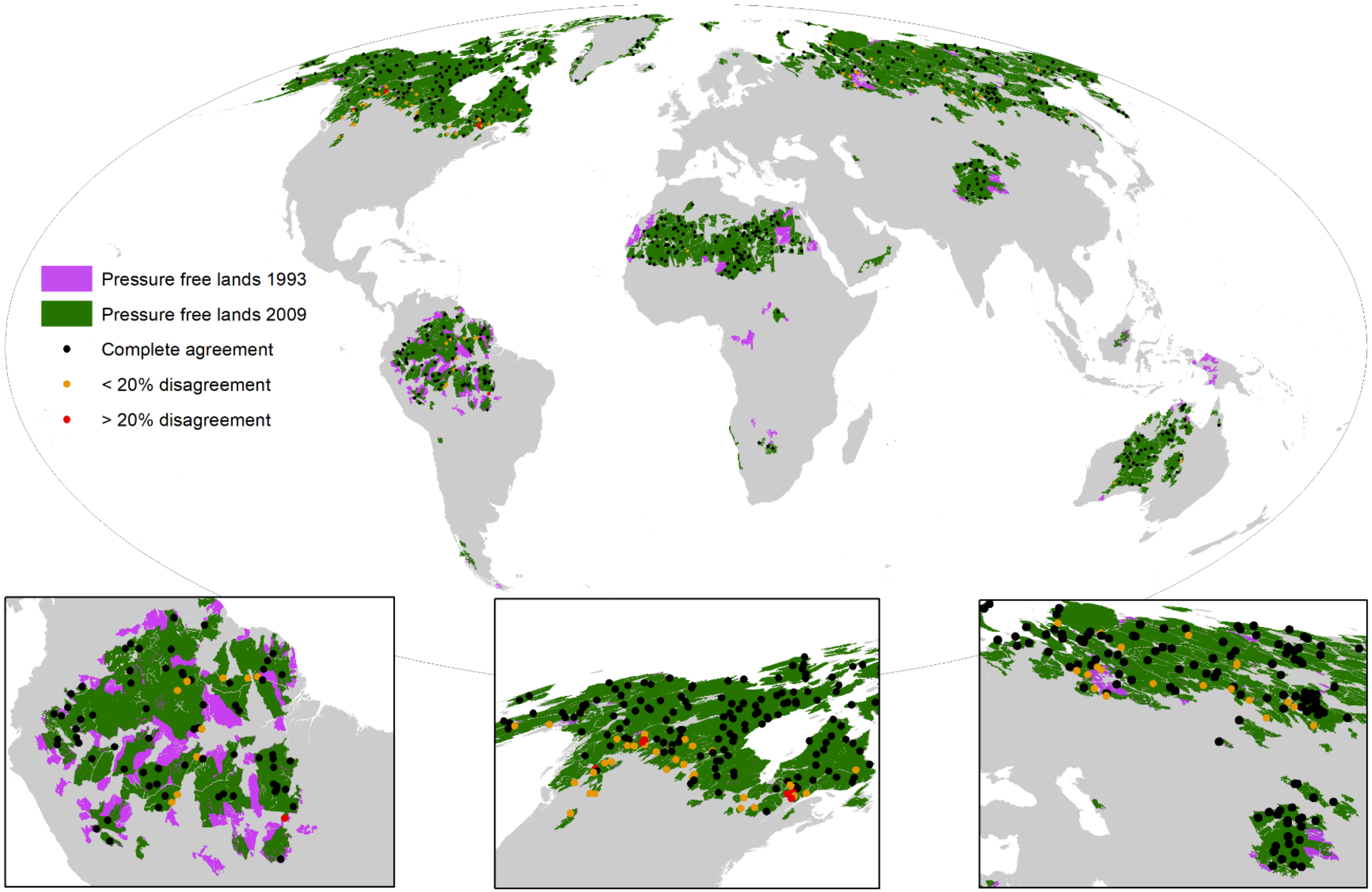
The Human Footprint dataset underpinning our wilderness mapping was the first cumulative pressure map to undergo data validation (Venter et al. 2016a). High resolution satellite imagery (ESRI World Imagery) was used to visually interpret human pressures in 3460 x 1km^2 plots across earth's terrestrial areas. A standard key for interpreting pressures was used and plots were also scored as certain or uncertain. Only plots where visual scores were certain (n = 3114) were used in the final validation exercise, and they had a median satellite imagery resolution of 0.5 meters. In general, a plot was scored as uncertain due to cloud cover or moderate resolution (15m) imagery. The Human Footprint score for each plot was determined through overlay in ArcGIS and both the visual and Human Footprint scores were normalised to a 0 – 1 scale making it possible to compare the two. Comparable imagery for 1993 was not available so only the 2009 map was validated.

The pressure scores in the visual validation and the Human Footprint strongly agree. The root mean squared error (RMSE) (Cort and Kenji 2005) and the Cohen kappa statistic of agreement (Viera and Garrett 2005) were used to determine Human Footprint performance. The RMSE is a dimensioned (expresses average error in the units of variable of interest) error metric for numerical predictions, and tends to heavily punish large errors. The RMSE was 0.125 on the normalised 0 -1 scale indicating an average error of approximately 13%.

The Kappa statistic expresses the agreement between two categorical datasets corrected for the expected agreement, which is based on a random allocation given the relative class sizes. When calculating the kappa statistic, the 2009 Human Footprint score was considered as a match to the visual score if they were within 20% (0.2 on 0-1 scale). The Kappa statistic was 0.737 ($P < 0.01$) which indicates strong agreement (Landis and Koch 1977, Viera and Garrett 2005). Of the visual validation plots 2757 (88.5%) were within 20% agreement. The Human Footprint scored 94 plots 20% higher than the visual validation score and 263 of them 20% lower. This suggests that the Human Footprint may be a slightly conservative measure of pressure, mapping pressures as absent in some places where they are actually present; however, the overall agreement is strong and encouraging. The sensitivity of the Kappa statistic to different thresholds for defining agreement was tested by Venter et al. (2016a) when validating the 2009 Human Footprint. With thresholds of within 15% and 25% the Kappa statistics were 0.565 (moderate agreement) and 0.856 (very strong agreement) respectively. This suggests some sensitivity but still shows good agreement.

To validate our map of pressure free lands in 2009 we identified all the plots from the Human Footprint visual validation which intersect our wilderness areas and assessed if they were in fact pressure free (Fig. 2.2). We used 624 plots with a median imagery resolution of 2.5 meters and found that 550 (88.1%) of the plots were scored through visual interpretation as completely free of human pressure. This shows strong agreement but suggests that in some places our maps are overestimating wilderness extent. We also found that 617 (98.9%) of the plots were within 20% (0.2) agreement of a Human Footprint score of zero on the 0-1 scale (pressure free) which is encouraging, and suggests that where we do overestimate wilderness the error is relatively small.

Figure 2.2 The extent of pressure free lands in 1993 (purple) and 2009 (green) with the results of the validation plots overlaid. Validation plots which were visually scored as pressure free and are therefore concordant with our definition of wilderness (Human Footprint = 0) are shown in black. Validation points that disagree by < 20% are shown in yellow, and those that disagree by > 20% are red.

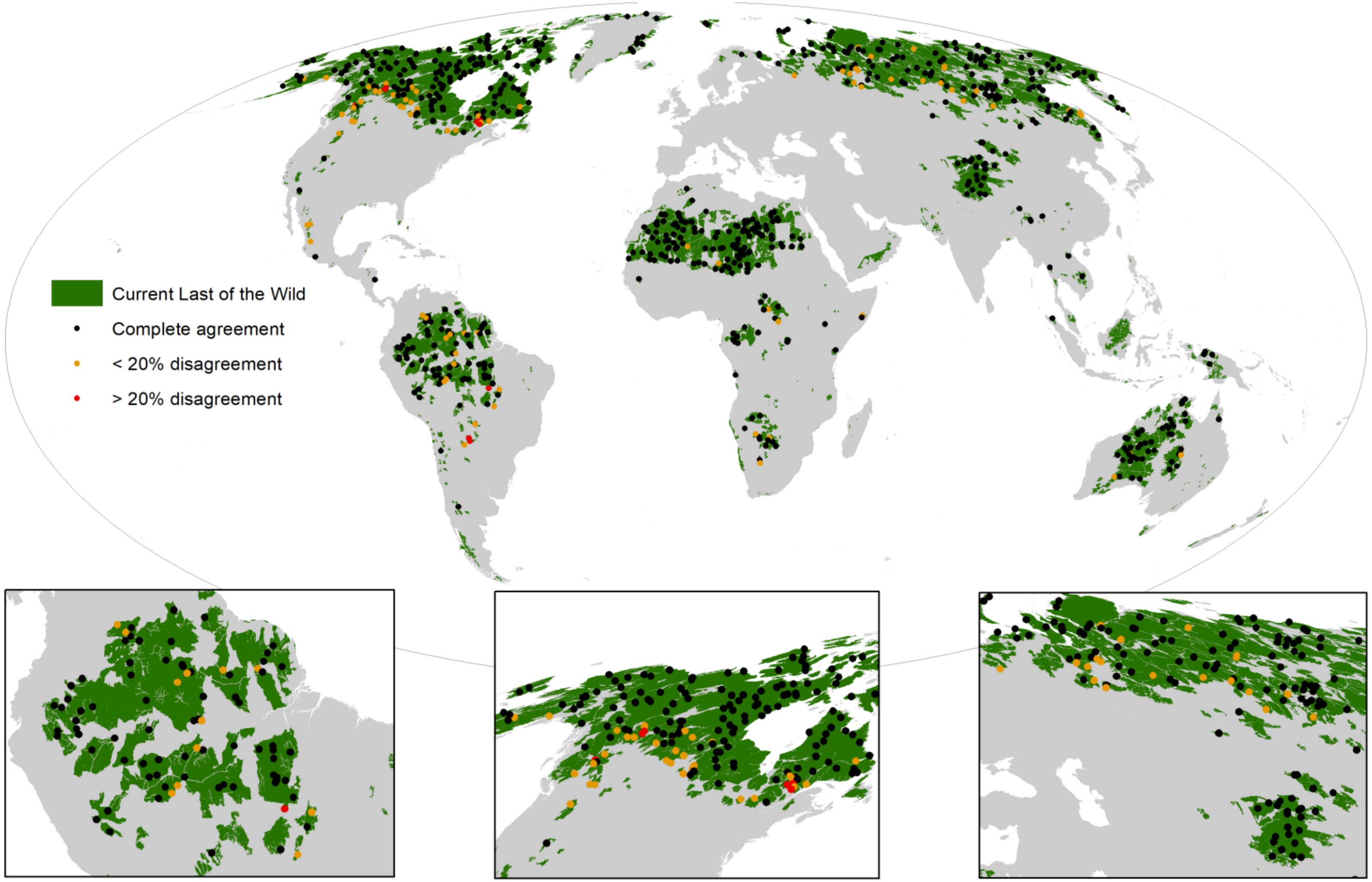


To validate the Last of the Wild map for 2009 we also identified all the plots from the Human Footprint visual validation which intersect those areas, and assessed if the standardised visual human pressure scores fell below standardised biorealm specific thresholds for the 10% area with the lowest Human Footprint in 2009 (Fig. 2.3). We used 687 plots with a median imagery resolution of 2.5 meters and found that 597 (86.9%) of the plots had visual pressure scores below their biorealm specific threshold showing strong agreement. If we consider scores up to 20% above a threshold as acceptable, then 678 (98.7%) of the plots are in agreement. Again, this suggests that the maps are overestimating wilderness in some places but that the errors are relatively small.

Usage Notes

The maps of wilderness we present are currently the most up-to-date products available. They are temporally inter-comparable, can support a range of analyses including monitoring changes in wilderness extent and fragmentation over time and are important information for conservation planning. The maps also include essential information needed to identify areas that could potentially meet the size and intactness criteria specified in the 2016 IUCN Global Standards for identifying Key Biodiversity Areas (IUCN 2016). Conserving wilderness areas is imperative for biodiversity conservation; as disturbance sensitive species disappear from human dominated landscapes, wilderness areas are becoming their last remaining strongholds (Gibson et al. 2011). These will be important sources of propagules and populations for restoration and re-wilding efforts, and serve as a baseline reference (Ceaușu et al. 2015, Pringle 2017).

Figure 2.3 The extent of the current Last of the Wild with the results of the validation plots overlaid. Validation plots which were visually scored as pressure free and are therefore concordant with our definition of wilderness (Human Footprint = 0) are shown in black. Validation points that disagree by < 20% are shown in yellow, and those that disagree by > 20% are red.



Protecting wilderness areas is also important because they provide high-value ecosystem services which are being lost in human modified and degraded landscapes (Klein et al. 2009, Freudenberger et al. 2012, Mackey et al. 2015, Martin and Watson 2016). Intact functioning ecosystems sequester and protect large amounts of carbon (Mackey et al. 2013), regulate local climate regimes including hydrological cycles (Bonan 2008, Pielke et al. 2011, Spracklen et al. 2012), and provide a direct defence against climate related hazards such as floods, sea-level rise and cyclones (The World Bank 2009). Protecting intact ecosystems is humanity's most cost effective defence against climate change (The World Bank 2009, Martin and Watson 2016), and may also prove to be the most cost effective way of meeting many of the United Nations Sustainable Development Goals (SDG's) (United Nations 2015c, Ibisch et al. 2016). The protection of wilderness areas could also serve as a direct indicator for progress towards certain SDG's, such as goal 15 which relates to biodiversity and ecosystem conservation (United Nations 2015c).

Many of the ecosystem services derived from wilderness areas are a direct result of their size, which allows them to act as complete self-organising systems (Sanderson et al. 2002). This has important implications for their conservation since damage in one area can affect the function of the entire system (Laurance 2005). For example, it is estimated that the Amazon needs 60% of its forest cover to retain its hydrological cycle (Sampaio et al. 2007). We anticipate our maps will be important tools for identifying places where conservation actions must occur at the ecosystem scale, and can help guide conservation efforts such as the implementation of mega-reserves (Laurance 2005).

The maps of wilderness we present have several important differences to other recently published products such as maps of intact forest landscapes (IFL's) (Potapov et al. 2017). IFL's are satellite-derived maps of the ecological state of the environment, whilst our wilderness maps are derived from maps of pressures or "threats". Pressures are actions which have the potential to damage nature, and therefore can drive changes in the ecological state of a system (Martins et al. 2012). Cumulative pressure maps such as the Human Footprint also combine top-down remotely sensed data and bottom up survey data to surmount the limitations of remotely sensed data such as lower accuracy in arid environments (Hansen et al. 2013b, Tropek et al. 2014, Venter et al. 2016c). Most importantly, our maps are not limited to a particular biome (e.g., forests), but rather span and consistently represent all non-Antarctic land areas.

Our work is subject to several caveats worthy of discussion. The Human Footprint relies on datasets which are globally comparable, but in some areas may not have the full extent of infrastructure that national or sub-national datasets contain or reflect all the pressures which could potentially impact on the wilderness quality of an area. For example, threats such as poaching, logging, forestry, invasive species, pollution and climate change are not directly captured, although many of them are often highly correlated to the pressures that were included in the Human Footprint (Venter et al. 2016c, a), such as human population density and road networks. There is a risk that the Human Footprint sometimes maps pressures as absent where they are actually present, underestimating human pressure in those parts of the world. In some cases, the human footprint also fails to account for historical land-use. For example, the island of Newfoundland in Eastern Canada has been clear-cut at least once, and lost wolves, its main predator over 100 years ago, yet appears as wilderness on our map. All of this suggests that our maps of wilderness are likely overestimates, and would benefit from being downscaled when used in a national or sub-national context (Tapia-Armijos et al. 2017).

CHAPTER 3 Hotspots of human impact on threatened terrestrial vertebrates

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Abstract

Conserving threatened species requires identifying where across their range they are being impacted by threats, yet this remains unresolved across most of Earth. Here we present the first high-resolution global map of cumulative human impacts on threatened species by using a spatial framework that jointly considers the co-occurrence of eight threatening processes and the distribution of 5,457 terrestrial vertebrates. The maps show that impacts are widespread, occurring across 84% of Earth's surface, and identify hotspots of impacted species richness, and coolspots of unimpacted species richness. Almost one quarter of assessed species are impacted across > 90% of their distribution, and ~7% are impacted across their entire range. These results foreshadow localised extirpations, and potential extinctions, without conservation action. The spatial framework developed here offers a tool for defining strategies to directly mitigate the threats driving species declines, providing essential information for future national and global conservation agendas.

Introduction

Human activities and land-uses are exerting unprecedented pressure on natural environments (Newbold et al. 2015, Venter et al. 2016c), threatening to drive tens of thousands of species to extinction (IUCN 2015). The main drivers of species declines include the conversion of natural habitats for land-uses such as crops, pasture and infrastructure, and the overexploitation of species through activities such as hunting (IUCN 2015, Maxwell et al. 2016). The distribution of these activities varies across Earth's terrestrial surface (Venter et al. 2016c), as do the distributions of the species they threaten (Jenkins et al. 2013). Understanding and quantifying spatial patterns of where human pressures overlap with sensitive species (i.e. mapping human impacts to threatened species) will improve our ability to prioritise actions to manage and mitigate human impacts on biodiversity (Wilson et al. 2006, Allan et al. 2013). Importantly, it will allow for the identification of areas across species distributions that are free from those threats which the species is sensitive to, and this information can be used to map global coolspots of what we call 'threat refugia'. Both forms of information are essential for conservation planning and can guide action towards securing these impact-free refugia, which are paramount for the survival of many threatened species (Hoffmann et al. 2010, Waldron et al. 2017).

Mapping impacts to biodiversity requires linking spatial data on the distributions of threats, with the distributions of species known to be sensitive to those threats (Halpern et al. 2008). To date, no efforts undertaken at either regional (Woolmer et al. 2008, Halpern et al. 2009) or global extents (Sanderson et al. 2002, Vorosmarty et al. 2010, Geldmann et al. 2014, Venter et al. 2016c, Ramírez et al. 2017) have accounted for the distribution and sensitivity of species and their threats, and therefore do not directly map likely human impacts (Martins et al. 2012). Past efforts that simply map threats (Venter et al. 2016c) fail to account for the distribution of species that respond to those threats, and even overlapping threats with species ranges (Evans et al. 2011) does not account for the specific sensitivities of each species to co-occurring threats. Some efforts to map threats to the marine realm estimated their impacts at the coarse ecosystem scale but did not account for individual species sensitivities (Halpern et al. 2008, Halpern et al. 2015). The few studies that do account for species have either been conducted at fine spatial resolutions (Bellard et al. 2015) or consider a limited number of taxonomic groups (Maxwell et al. 2013, Shackelford et al. 2018), and many suffer from the assumption that species are exposed to threats across their entire range, not just where the threat occurs, overestimating impacts (Schipper et al. 2008, Evans et al. 2011, Moran and Kanemoto 2017). Clearly our understanding of where individual species are being impacted by threats, or where their threat-free refugia are, remains limited at the global scale (Joppa et al. 2016), and is a major gap in our ability to prioritise conservation actions (Tulloch et al. 2015b, Joppa et al. 2016).

Here, we present the first global assessment of the spatial distribution of human impacts on globally threatened and near threatened terrestrial birds, mammals and amphibians. We developed a novel method for quantifying and mapping human impacts that jointly considers the distributions of 5,457 threatened and near threatened species (1,277 mammals, 2120 birds, and 2060 amphibians), and the distribution of species-specific threats, and the extent to which the distribution of each species is impacted by relevant threats (Fig 3.1).

Spatial data on threats was obtained from the recently updated Human Footprint (Venter et al. 2016c), which is unique for considering eight human pressures globally at a 1km^2 resolution, including: built environments, crop lands, pasture lands, human population density, night lights, railways, major roadways and navigable waterways. This makes the Human Footprint the most complete and highest resolution globally consistent dataset of anthropogenic threats (McGowan 2016). Each individual pressure was linked to a species if they directly or indirectly correspond to threats identified by the IUCN Red List (IUCN Downloaded on 12 December 2015) as driving the endangerment of that species. The Human Footprint data correspond with seven major classes, and 15 sub-classes of IUCN threats (Table 3.1). Although these do not include all threats to species, they do include all of the most prevalent drivers of global biodiversity decline (Maxwell et al. 2016). We calculated the proportion of each species range that is currently impacted by a threat, and then mapped cumulative human impacts in a 30 km x 30 km grid globally (see Methods). We also examined patterns of human impacts across individual species distributions, taxonomic groups and threat status categories. Finally, we used the inverse of our cumulative impact maps to identify threat refugia, the places where high numbers of threatened (and near threatened) species persist unimpacted by human activity.

Figure 3.1 Methodological framework for mapping cumulative human impacts on threatened vertebrate species.

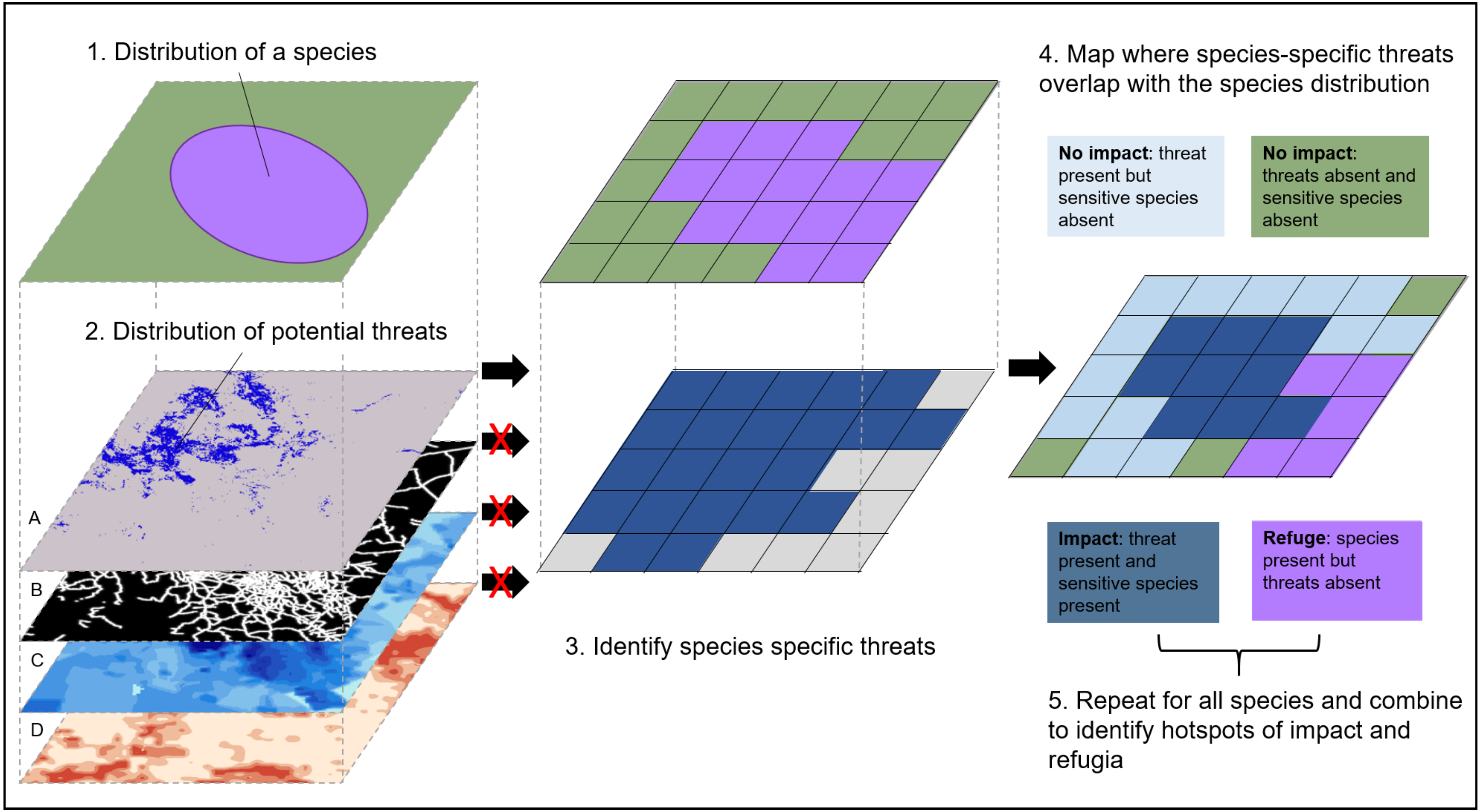


Table 3.1 Major classes and sub-classes of threats to biodiversity, as classified in the IUCN Red List of Threatened Species, and the corresponding spatially explicit pressure variable from the updated Human Footprint dataset.

Major threat class (IUCN)	Sub-class threats (IUCN)	Pressure (Human Footprint)	Species Impacted
1. Residential & commercial development	1.1 Housing & urban areas	Electric infrastructure (nightlights) Built environments	1748
	1.2 Commercial & industrial areas	Electric infrastructure (nightlights) Built environments	349
2. Agriculture & aquaculture	2.1 Annual and perennial non-timber crops	Crop lands	4017
	2.3 Livestock farming & ranching	Pasture lands	1850
4. Transportation & service corridors	4.1 Roads & railroads	Railways Roads	563
	4.2 Utility &service lines	Roads	88
5. Biological resource use	5.1 Hunting and collecting terrestrial animals	Navigable waterways Population density Roads	1594
	5.2 Gathering terrestrial plants	Navigable waterways Population density Roads	149
6. Human intrusions & disturbance	6.1 Recreational activities	Electric infrastructure (nightlights) Population density	373

	6.3 Work & other activities	Electric infrastructure (nightlights) Population density	196
8. Invasive & other problematic species, genes & diseases	8.1 Invasive non-native / alien species / diseases	Population density Roads	1319
9. Pollution	9.1 Domestic and urban waste water	Population density Built environments	205
	9.3 Agriculture & forestry effluents	Crop lands	805
	9.4 Garbage & solid waste	Built environments	27
	9.6 Excess energy	Electric infrastructure (nightlights) Built environments	24

Results

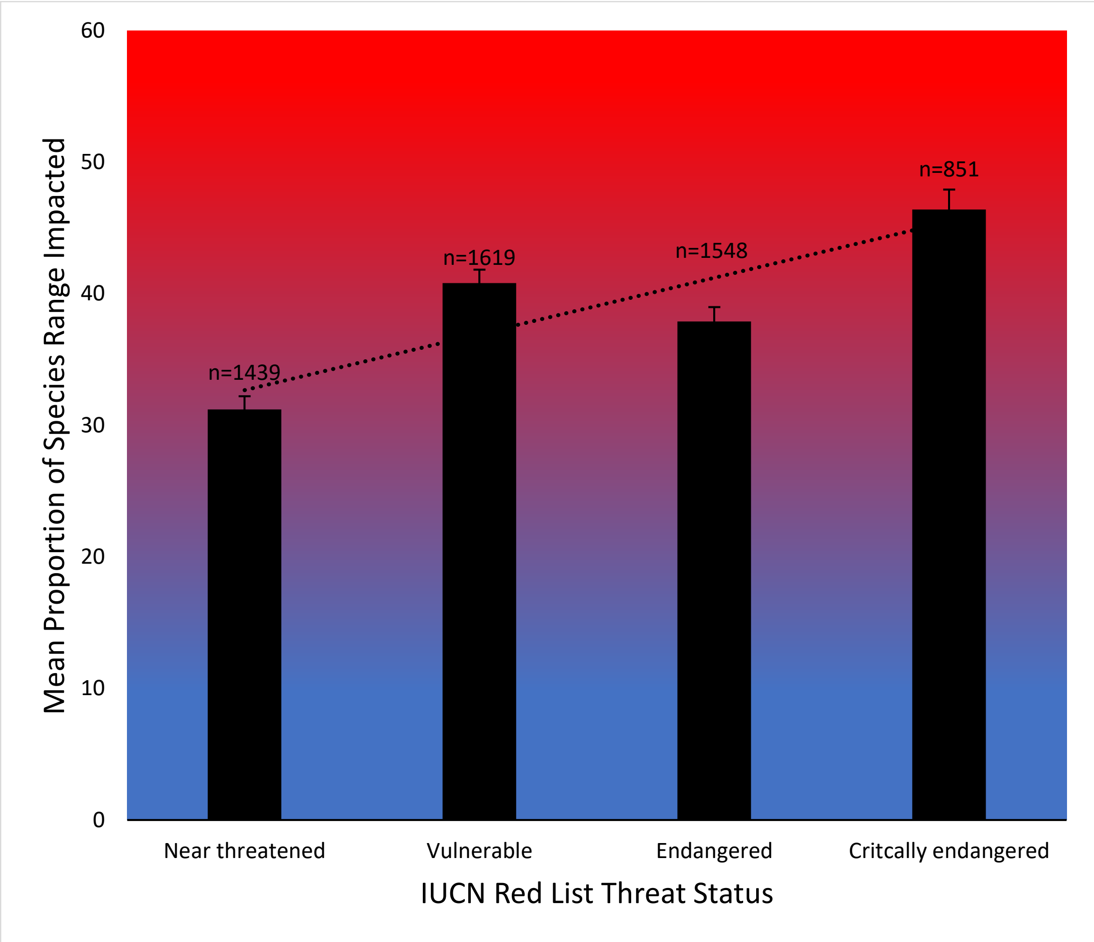
Human impacts on threatened vertebrate species

We found that on average 38% of a species' distribution range is impacted by one or more relevant threats (Table 3.2), including an average 21% of the distribution impacted by multiple co-occurring threats. Mammals are the most impacted of all taxa, with on average 52% of a species' distribution impacted by relevant threats. Concerningly, almost one quarter of all species (23%, n=1237) are impacted by threats across >90% of their distribution, with 395 (7%) impacted by at least one relevant threat across their entire distribution. Conversely, we found that one third of all species (34%, n=1863) are not exposed to the threats we mapped across any portion of their distribution; however, this result should be interpreted within the context of threats we consider. We found that the proportion of a species distribution impacted by threats is correlated with its threat status (IUCN Red List categories; Fig. 3.2) (Analysis of variance $P < 0.001$, $F = 7.5$). Species classified as critically endangered on the IUCN Red List had almost half their distribution impacted by threats on average (46%, n=851), whilst near threatened species had one third of their distribution impacted by threats on average (31%, n=1439).

Table 3.2 The number (and percentage) of species and the proportion of their distribution impacted by threats.

	Total number of species	100% impacted	>90% impacted	>50% impacted	0% impacted	Mean proportion impacted (%)
Amphibians	2060	171 (8.3%)	384 (18.6%)	685 (33.3%)	1082 (52.5%)	31.5
Birds	2120	88 (4.2%)	380 (17.9%)	822 (38.8%)	387 (18.3%)	37.2
Mammals	1277	111 (8.7%)	465 (36.4%)	681 (53.3%)	337 (26.4%)	51.5
Total	5457	370 (6.8%)	1229 (22.5%)	2188 (40.1%)	1806 (33.1%)	38.4

Figure 3.2 Mean proportion of species distributions impacted by threats across extinction risk categories of threatened and near threatened terrestrial vertebrates. Bars represent means with standard errors. Species extinction risk assessed by the International Union for Conservation of Nature (IUCN 2015).



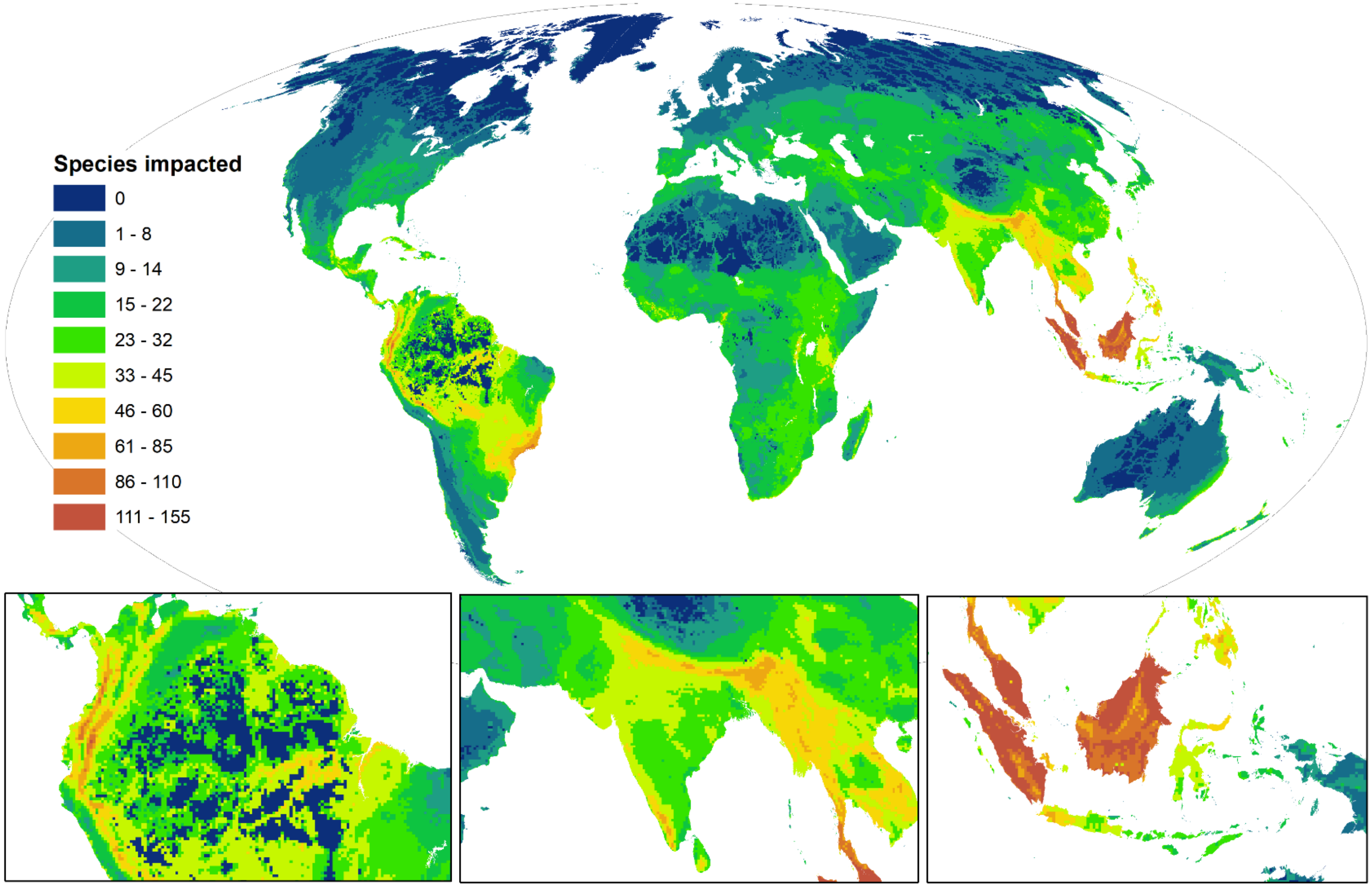
Global hotspots of human impact

Human impacts on threatened vertebrates are widespread, extending across 84% of Earth's terrestrial surface (Extended Data Table 3.1; Extended Data Fig. 3.1-3.2). There is strong spatial variation in the intensity of human impacts, with alarming peaks in Southeast Asia (Fig 3.3). Hotspots of human impact differ spatially between taxa (Extended Data Fig 3.3.), and as expected, are largely driven by patterns of threatened species richness (Extended Data Fig. 3.4) and human pressure, although they are not congruent.

The top five countries most impacted by anthropogenic threats to species are all found in Southeast Asia (Extended Data Table 3.2), which we confirm is overwhelmingly the dominant global hotspot of impacts to species (Sodhi et al. 2004). Malaysia has the highest average human impact score (125 species impacted per grid cell), followed by Brunei and Singapore (124 and 112 species respectively). These scores are substantially higher than the global average of 16 species impacted per grid cell. Concerningly, there are 13 grid cells (11,700km^2) in Southeast Asia where >150 species are impacted by threats.

When aggregated across biomes and ecoregions, which represent distinct biogeographic spatial units at the global scale (Olson et al. 2001) (Extended Data Table 3.3), the highest human impacts are in *Mangroves*, where on average 35 species are impacted per grid cell. Human impacts are also high throughout the tropical forests which harbour Earth's richest biota, and are critically important for biodiversity conservation (Gibson et al. 2011). The *Tropical and sub-tropical moist broadleaf forests* in Southeast Brazil, Malaysia, and Indonesia are the second most impacted biome, followed by the *tropical and subtropical dry broadleaf forests* in India, Myanmar, and Thailand (35 and 34 species impacted per 900km^2 grid cell).

Figure 3.3 Cumulative human impacts on threatened and near threatened terrestrial vertebrates (n=5457). Legend indicates the number of species in a grid cell impacted by at least one threat. Areas of high human impact (hotspots) are shown in Red. Maps use a 30x30 km grid and a Mollweide equal area projection.



Global coolspots of threat refugia

We mapped threat refugia for threatened vertebrates by combining the unimpacted parts of each species' distribution (Fig. 3.4). Almost the entire Earth's surface (97%) host at least one unimpacted threatened species, acting as a potential refugium for that species (Fig. 3.4); however, impacted and unimpacted species co-occur across 80% of Earth's surface, identifying places where species with divergent sensitives to threatening processes are present. There is strong spatial variation in the intensity of threat refugia for threatened species, and between coolspots for different taxa (Extended Data Fig. 3.5). Threat refugia often follow similar patterns to hotspots of impact, with Southeast Asia again the dominant global hotspot. Although counterintuitive, our results are largely driven by species richness and individual species different sensitivities to threats. Therefore, in species rich areas it is logical that many species will be impacted, whilst many others remain unimpacted. The highest average threat refugia score is in Brunei (49 species unimpacted per grid cell), but the highest score for an individual grid cell occurs in Malaysia, where 144 species are unimpacted. Encouragingly, there are 12 grid cells (10,800km^2) in Southeast Asia with >100 unimpacted species, although this is primarily due to the large number of threatened species in the region.

Other coolspots of threat refugia include Liberia in West Africa, the Amazon rainforest and Andes mountains in South America, and the Eastern Himalayan biodiversity hotspot in Nepal, Bhutan and Myanmar. When aggregated across Biomes and ecoregions (Extended Data Tables 3.3), the *Tropical and sub-tropical moist broadleaf forests*, and *tropical and subtropical dry broadleaf forests* act as the greatest threat refugia supporting on average 29 and 22 unimpacted species per grid cell respectively. These are also two of the most impacted biomes, demonstrating that despite this, there are still considerable conservation opportunities here. The *Tundra* and *Boreal forest* are the only Biomes where more species are unimpacted than impacted on average.

Proportion of species impacted

Some areas of the planet contain low numbers of threatened species (e.g. the high latitudes, or arid and desert regions). Therefore, it is instructive to examine the corresponding proportions of impacted versus unimpacted species. On average there are more impacted than unimpacted species in a grid cell globally (15.6 versus 13.8; ratio 1.13) (Fig. 3.5)). The proportion varies for taxonomic groups, with Amphibians having the highest ratio of impacted versus unimpacted species (2.3 versus 1.6; ratio 1.5), compared to birds and mammals (Birds 10.5 versus 9.3; ratio 1.2 & mammals 5.4 versus 5.1; ratio 1.1).

In our 30 km^2 grid cells, the proportion of species impacted extends across the full range from 0 – 100%. We found that > 90% of species were impacted in 3,826 grid cells globally, amounting to a staggering 3.4 million km^2 (2.4% of Earth's terrestrial area), which is an area greater in size than India. Encouragingly, species are present but none are impacted in 24,233 grid cells (21.8 million km^2; 15.1% of Earth's terrestrial area). The majority of this is wilderness where no human pressures occur. However, we found 919 grid cells (827,100 km^2; 0.5% of terrestrial area) where a species and a human pressure co-occur, but there is no impact (i.e. the species is not sensitive to the human activity or land use occurring in that area).

The distribution of areas with high proportions of impacted species differs substantially from hotspots of human impact. Europe and North and Central America now emerge as global hotspots, particularly for mammals and amphibians. The proportion of birds impacted presents a more spatially homogenous pattern, with hotspots in Southeast Asia and the Southeast South America. When aggregated across biomes, *Mangroves* have the highest mean proportion of impacted species (61.3%), followed by *Temperate broadleaf and mixed forests* (60.7%) (Extended Data Table 3.3). The *Tundra* and *Boreal/taiga forests* have the lowest mean proportions of impacted species (14.6% and 29% respectively).

Figure 3.4 Hotspots of refugia for threatened and near threatened terrestrial vertebrates (n=5457). Legend indicates the number of species that are not impacted by any threats in a grid cell. Hotspots of refugia are shown in yellow/green. Maps use a 30x30 km grid and a Mollweide equal area projection.

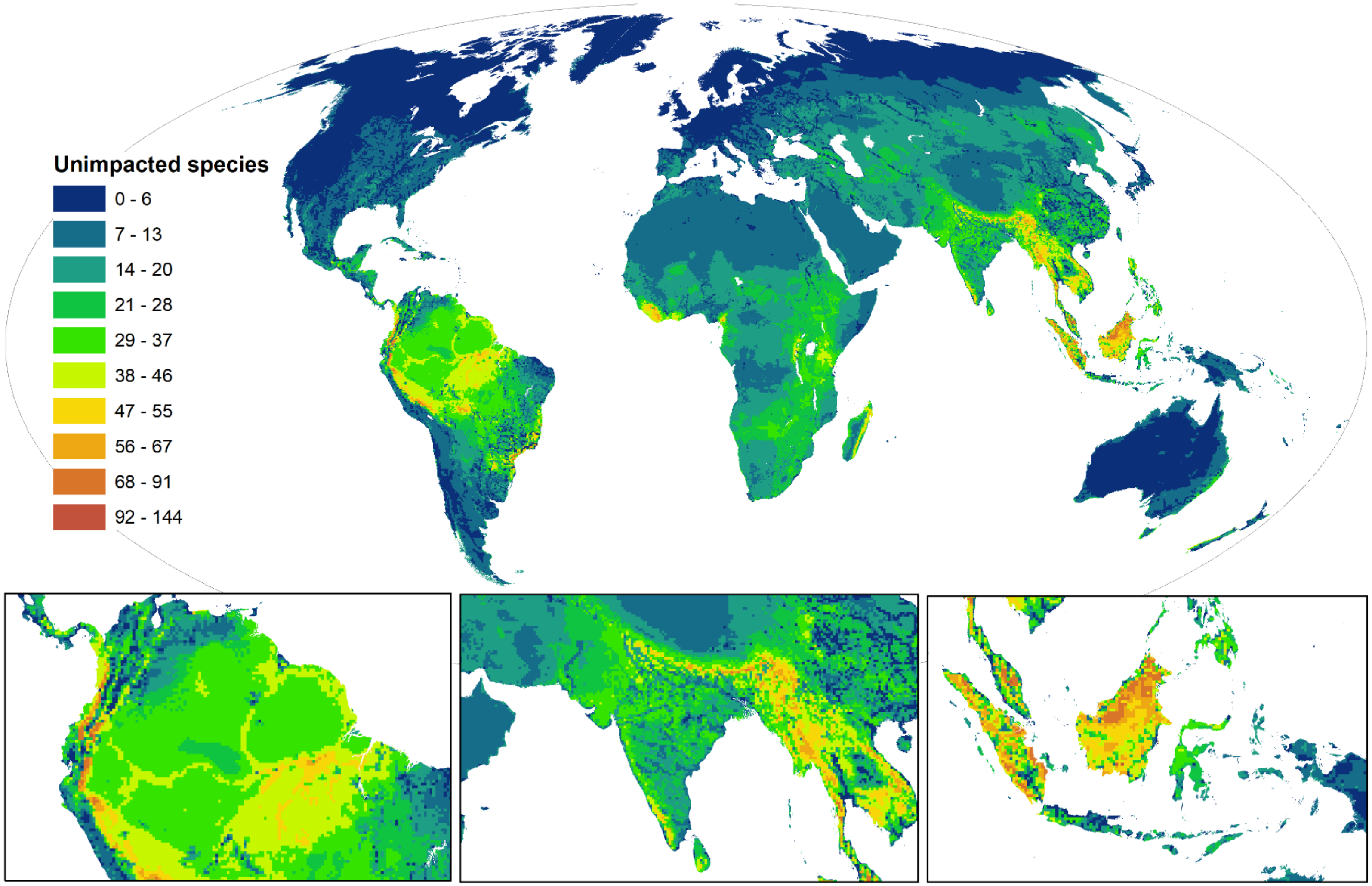
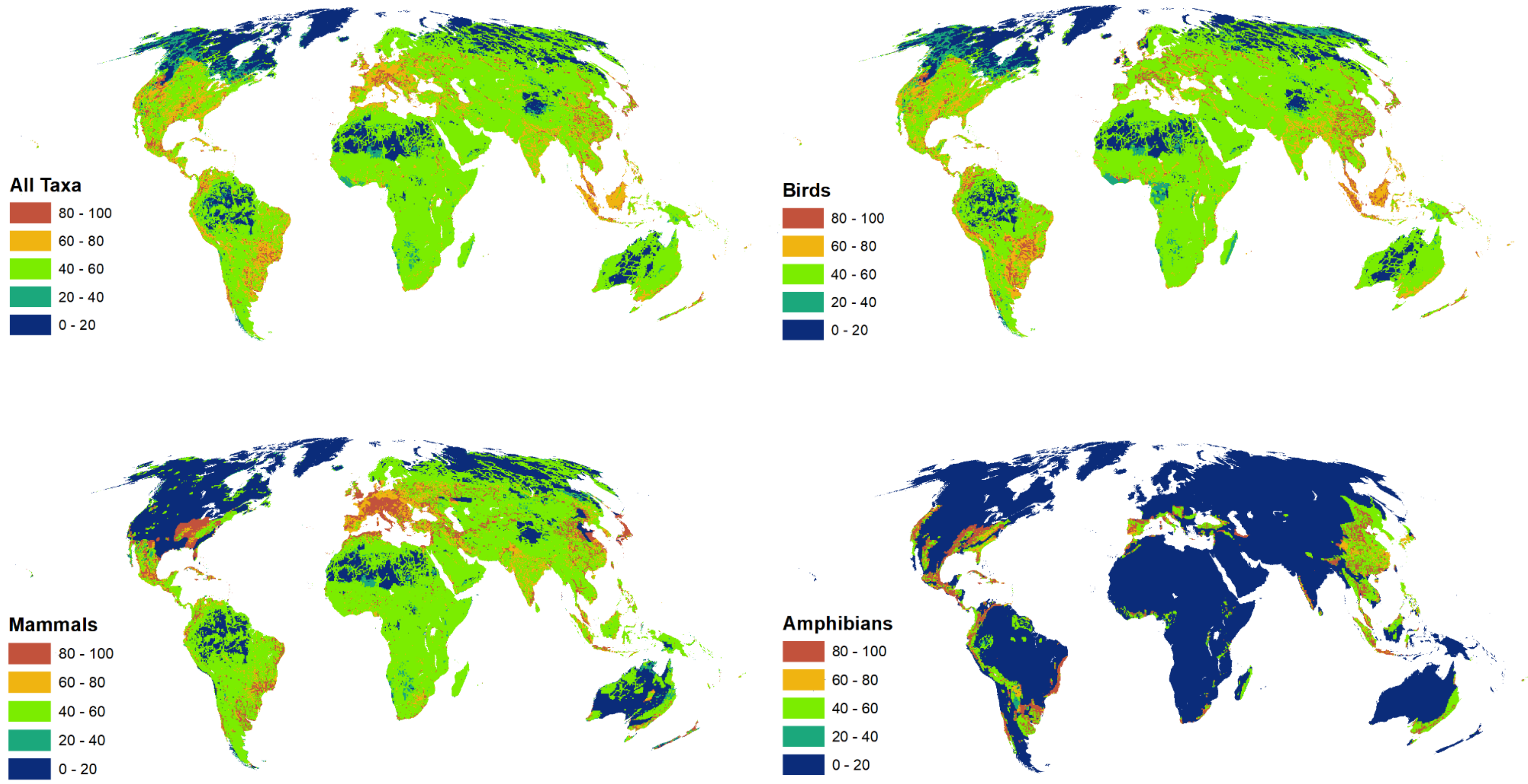


Figure 3.5 The proportion of species in a grid cell impacted by a threat (and inversely the number of unimpacted species for whom it is a refuge) for all taxa (n=5457), mammals (n = 1277), birds (n = 2120), and amphibians (n=2060). Maps use a 30x30 km grid and a Mollweide equal area projection.



Discussion

Implications for biodiversity conservation

Our results represent the current best estimate of the spatial distribution of human impacts on terrestrial vertebrates. Continued extirpations, the precursors of extinction, will continue to occur in the impacted portions of species ranges, which our results demonstrate are substantial. Consequently, completely impacted species, or those persisting in threat refugia that are too small to support viable populations in the long term (Maron et al. 2012), likely face imminent extinction. These findings complement recent work showing that hundreds of mammals have lost considerable portions of their historic distributions (Ceballos et al. 2017), and that habitat fragmentation has greatly reduced the proportion of highly suitable habitat within species distributions, reducing their movements (Tucker et al. 2018), and increasing their extinction risk (Crooks et al. 2017). Clearly, the current anthropogenic erosion of biodiversity is more severe than previously thought.

Although our results are concerning, there is room for hope. The threats we map can be mitigated by *in situ* conservation actions, but diverse approaches are required. To ensure the survival of highly impacted species with little or no threat refugia, active threat management, restoration and rewilding efforts (Ceaușu et al. 2015) are needed to open up enough viable habitat for species to persist. Conservation action in the hotspots of human impact we identify will have high benefits since they are areas with exceptionally high threatened species richness and species-specific threats (Myers et al. 2000). Our results therefore extend previous efforts to identify biodiversity hotspots (Myers et al. 2000), which were developed following somewhat similar logic, and have helped guide conservation action and millions of dollars of funding. The hotspots of human impact we identify are priorities for actions that mitigate the specific threats (Brooks et al. 2006).

Rather than being purely reactive and focusing solely on securing a future for imperilled species in the short term, conservation efforts would also benefit from proactively securing coolspots of species refugia and avoiding any initial human impacts in these places (Betts et al. 2017). This will guarantee the persistence of many species in the long term, especially in a time of a rapid climate change, where areas free of threatening processes will be critical for species adaptation (Martin and Watson 2016, Scheffers et al. 2016). Securing threat refugia will be particularly effective if protection is targeted at the most species rich places that currently remain threat free but may soon be jeopardised (Venter et al. , Margules and Pressey 2000). Additionally, conservation action is also likely to have a high chance of success in threat refugia and also be more cost-effective (Balmford et al. 2003, Tulloch et al. 2015a). Proactive and reactive approaches to conservation have historically been pitted against each other (Kareiva and Marvier 2003), with reactive approaches deemed more urgent and taking precedence (Kareiva and Marvier 2003, Hoekstra et al. 2005, Pressey et al. 2017). However, our discovery of the spatial overlap existing between hotspots of impacted species richness and coolspots of unimpacted species richness provides opportunities for multi-faceted conservation action that is reactive for some species and while simultaneously being proactive for others.

The utility of our work extends beyond conservation and can also inform sustainable development planning. Conservation action within some of the hotspots we identified (especially in Southeast Asia) are likely to deliver synergistic benefits to other environmental goals, such as carbon conservation and global reduction of deforestation rates (Di Marco et al. 2016a). Additionally, according to our definition, species threat refugia do not necessarily have to be off limits to human development, just free of the actions and land-uses that directly threaten species found in that area. This provides a unique framework for quantifying the trade-offs associated with the development of alternate human activities and land-uses, and for identifying locations and strategies to minimise their impacts on biodiversity. This has implications for nations striving to meet ambitious development targets such as the United Nations Sustainable Development Goals (SDGs), especially where achieving development goals involves trade-offs with biodiversity goals (Ibisch et al. 2016, Singh et al. 2017). The framework presented here could be adapted to inform conservation and development planning from local to regional scales, and could be particularly useful in South East Asia, Latin America and sub-Saharan Africa; regions that are undergoing rapid economic development, but are also hotspots of human impact and coolspots threat refugia (Laurance et al. 2009, Wadey et al. 2018).

It is important to note that our data are not comprehensive of all threats to all species. For example, our analysis does not take into account infectious diseases, a major driver of global declines in amphibians (Stuart et al. 2004), or climate change, a threat already impacting many species across all taxa (Scheffers et al. 2016). The results are therefore conservative, and many species will be more impacted than our maps indicate. Notably, one of the fundamental ways to manage global scale threats such as climate change, is to stop more easily abatable threats such as those considered in this analysis (Ripple et al. 2016), and to avoid antagonistic or synergistic interactions between multiple threats (Brook et al. 2008, Mantyka-Pringle et al. 2015). Other caveats worthy of discussion are that we assume the intensity of a threat (e.g. agricultural land use or roads) are equal across their distribution, and that species are equally sensitive to each threat known to affect them. This assumption could mean we are overestimating impacts in cases where species are sensitive to several threats where only the secondary threat is present. The IUCN has collected data on the severity of threats to species, but a comprehensive database is still lacking as this information is often unknown. The further development of these data would allow important nuances to be included in future extensions of this work.

A species and threat overlap does not necessarily mean that the threat is acting in that location. However, our analysis extends beyond a species threat overlay by incorporating three co-occurring and connected forms of data; a species distribution, a threat distribution, and that species vulnerability to that threat. To the best of our knowledge, this is the first-time species-specific sensitivity to threats has been incorporated into an impact mapping exercise at this scale. By mapping species-specific threats, it is much more likely that a threat is acting in a given location and impacting a species. This approach does rely completely on the current knowledge of threats to species, and assumes no other currently undocumented threats could be impacting a species. We sourced information on threats to species from the IUCN, who are the main authority on assessing species extinction risk, and limited our analyses to threatened terrestrial vertebrates, which include the most studied taxa globally (Di Marco et al. 2017). Yet, it is important to note that there is still variation between species assessments due to taxonomic and geographical biases which could influence our findings (Donaldson et al. 2016). For example, our understanding of threats to mammals is greater than for amphibians, which could partly explain why our results show mammals as the most impacted taxon, whilst amphibians are generally regarded as the more threatened taxon.

This analysis provides a framework for mapping human impacts that represents a conceptual advance over cumulative pressure mapping or threatened species richness mapping that can be applied to any scale, taxa or realm. Furthermore, the framework and baseline can be continually updated and enhanced as additional data on species distributions, their sensitivity to threats, and the spatial distribution of threats become available, and our understanding of threat interactions improves. Improvements in our understanding of species sensitivity to threats will also allow this analysis to be extended to other forms of life such as plant and invertebrate species. We have shown that human impacts on species are almost ubiquitous across Earth, and that hundreds of species have no refuge from these impacts, including many of the most charismatic large mammals. The survival of these species, and many more, hinges on humanity’s ability and willingness to compromise and share space.

Methods

Spatial data on threatened species ranges

We focused our analysis on terrestrial vertebrate groups (amphibians, birds, mammals) with distribution maps and assessment of identified threat available for all species. Spatial data on mammal and amphibian distributions was obtained from the IUCN Red list of threatened species (IUCN 2015), and bird distributions from Birdlife International and NatureServe (NatureServe 2015). We focused on species which are listed as near threatened, vulnerable, endangered or critically endangered since their major threats have been identified and comprehensively assessed for the IUCN Red List of Threatened Species (Rodrigues et al. 2006, IUCN 2015, Brooks et al. 2016, Maxwell et al. 2016). Threats to species could be operating in the past, ongoing and/or likely to occur in the future (IUCN Downloaded on 12 December 2015). Following established practice we only considered native and reintroduced parts of each species distribution range in our analysis, which are listed as extant, possibly extant or possibly extinct within their range (Butchart et al. 2015). We excluded introduced, vagrant and extinct species as well as species whose origin or presence is uncertain. Although reintroduced species ranges may be theoretically subject to less threats, they may still be under threats not realised during the reintroduction process (Seddon et al. 2014). As such, incorporating all portions of a species range, including reintroduced areas, can provide a robust picture of the threats for a given species. Finally, we only included species whose distribution overlapped (even just partially) with the extent of the Human Footprint threat dataset, which does not include Antarctica. A total of 2060 amphibian species, 2120 bird species, and 1277 mammal species qualified for our analysis based on these criteria.

Spatial data on threats to species

Spatially explicit data on the distribution of threats to species was obtained from the recently updated Human Footprint maps (Venter et al. 2016b, Venter et al. 2016a). These are globally standardised maps of cumulative human pressures on the natural environment at 1km^2 resolution globally for eight of the most harmful pressures humans exert on nature including: 1) built environments, 2) population density, 3) electric infrastructure, 4) crop lands, 5) pasture lands, 6) roads, 7) railways, and 8) navigable waterways. This makes the Human Footprint the most up-to-date and comprehensive global cumulative pressure/threat map available (McGowan 2016). The Human Footprint is also the first global scale threat dataset to have been validated for accuracy. This was done by visually confirming if human pressures were present or absent across thousands of randomly selected 1km x 1km plots globally (Venter et al. 2016a). The data were found to exhibit an excellent degree of accuracy (88.5% agreement between visual plots and human footprint data), especially at identifying threat free areas (98.9% agreement between visual plots and wilderness) (Allan et al. 2017d).

In the Human Footprint each pressure layer is scaled between 1 and 10 based on its estimated impact on the environment. These scores are then cumulated in each pixel to give a total score out of 50. We converted these scores to binary (present or absent in any 1km^2 pixel) for our analyses since there is not data on the relative severity of individual threats to species. To convert pressure layers from continuous scales to binary (present/absent) we set cut-offs where the pressure was considered absent. For example, roads have a direct pressure score of 8 up to 500 meters either side, beyond this the pressure score decays exponentially from a score of 4 out to 0 at 15km. When converting this to a binary score, we set a threshold that considered the pressure present up to 3 km either side of the road, and absent beyond this (see Extended Data Table 3.4 for comprehensive details on how each layer was handled).

Mapping species-specific threats

We identified cases where the eight pressures in the updated Human Footprint dataset directly or indirectly correspond with threats to biodiversity as listed in the IUCN Red List (IUCN Downloaded on 12 December 2015) (Table 3.1.). This allowed us to globally map seven major classes, and 15 sub-classes of threats. Although this is not comprehensive of all the threats to species, it importantly includes the biggest drivers of biodiversity declines globally (Maxwell et al. 2016). For example multiple forms of agriculture, urban development and transportation corridors are directly accounted for by our pressure data. Whilst biological resource use and over-exploitation through hunting, pollution, human disturbance, and invasive species are indirectly accounted for by human population density, roads and navigable river networks which act as proxies (Hulme 2009, Laurance et al. 2009, Meunier and Lavoie 2012, Laurance et al. 2015, Ripple et al. 2016).

Analysing the extent of human impacts on individual species

For a pressure to impact a species, it must spatially overlap with that species' distribution, and have been identified in the IUCN Red List as a threat to that species(Martins et al. 2012). Therefore, we calculated the extent of overlap between each species distribution, and each pressure layer which that species is sensitive to at a $1km^2$ resolution globally. We accounted for the overlap between threats, identifying where multiple threats are present. All spatial data was analysed in a Mollweide equal area projection in ESRI ArcGIS and PostGIS, and statistics were calculated in R statistical software. We used a one-way analysis of variance to test for correlation between a species extinction risk category and the proportion of that species range impacted by threats.

Mapping hotspots of cumulative human impacts

We estimated cumulative human impacts on threatened species using a global 30km x 30km planning unit grid, since this has been identified as the ideal resolution for reducing the effects of commission errors (where species are thought to be present but are not) when working with species range maps (Di Marco et al. 2016c). An impact was scored in a grid cell if a species and at least one threat it is sensitive to were both present. This means that the presence of a threat and a species in the same grid cell is not considered an impact unless the species is known to be sensitive to that threat. We then calculated the sum of all impacted species in a grid cell to give a total estimate of cumulative human impact.

As a sensitivity analysis, we calculated the area of each species distribution within each planning unit and the area of each pressure in each planning unit, converting both to proportions of planning unit area. To estimate how impacted each species is within each planning unit, we multiplied the proportion of the species distribution by the proportion of each pressure which threatens and then summed the scores. By using the proportion of planning unit area, we scale for the likelihood of a species and a pressure overlapping within a grid cell. Finally, we calculated the sum all the individual species impact scores within each grid cell, to give a total estimate of cumulative human impact. Spatial patterns of impact were strongly coherent between the two approaches so we report on the more intuitive binary metric in the manuscript.

Mapping coolspots of threatened vertebrate anthropogenic refugia

We followed similar methods to mapping human impacts, where a cell was scored as an anthropogenic refuge if a species was present in the cell, but no pressures that threaten it were present. These were then summed to give a cumulative score of the number of unimpacted species in a cell.

CHAPTER 4 Recent increases in human pressure and forest loss threaten many Natural World Heritage Sites

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Abstract

Natural World Heritage Sites (WHS), via their formal designation through the United Nations, are globally recognized as containing some of the Earth's most valuable natural assets. Understanding changes in their ecological condition is essential for their ongoing preservation. Here we use two newly available globally consistent data sets that assess changes in human pressure (Human Footprint) and forest loss (Global Forest Watch) over time across the global network of terrestrial WHS. We show that human pressure has increased in 63% of WHS since 1993 and across all continents except Europe. The largest increases in pressure occurred in Asian WHS, many of which were substantially damaged such as *Manas Wildlife Sanctuary* and *Simien National Park*. Forest loss occurred in 91% of WHS that contain forests, with a global mean loss of 1.5% per site since 2000, with the largest areas of forest lost occurring in the Americas. For example *Wood Buffalo National Park* and *Río Plátano Biosphere Reserve* lost 2581km^2 (11.7%) and 365km^2 (8.5%) of their forest respectively. We found that on average human pressure increased faster and more forest loss occurred in areas surrounding WHS, suggesting they are becoming increasingly isolated and are under threat from processes occurring outside their borders. While some WHS such as the *Sinharaja Forest Reserve* and *Mana Pools National Park* showed minimal change in forest loss or human pressure, they are in the minority and our results also suggest many WHS are rapidly deteriorating and are more threatened than previously thought.

Introduction

The World Heritage Convention was adopted in 1972 to ensure the world's most valuable natural and cultural resources could be conserved in perpetuity (UNESCO 1972). The Convention aims to protect places with Outstanding Universal Value that transcend national boundaries, and are worth conserving for humanity as a whole. These places are granted World Heritage Status, the highest level of recognition afforded globally (UNESCO 2015). A unique aspect of The Convention is that host nations are held accountable for the preservation of their World Heritage Sites by the international community, and must report on their progress to the United Nations Educational, Scientific and Cultural Organisation (UNESCO). Over 190 countries are signatories to The Convention, committing to conserving the 1031 World Heritage Sites listed at the time of this study (UNESCO 2015). Of these, 229 are Natural World Heritage Sites (WHS), inscribed for their unique natural beauty and biological importance, including many of the world's most important places for biodiversity conservation such as the *Pantanal Conservation Area* in Brazil (UNESCO 2016a) and the iconic *Serengeti National Park* in Tanzania (UNESCO 2016c).

As the number of WHS has increased over the last few decades, so have the pressures humanity is exerting on the natural environment (Rockstrom et al. 2009, Steffen et al. 2015a, Venter et al. 2016c). Anthropogenic habitat conversion due to human activities such as agriculture and urbanisation are driving biodiversity extinction rates well above background levels, and the condition of many ecosystems is in decline worldwide (Barnosky et al. 2012, Hansen et al. 2013b, Pimm et al. 2014, Watson et al. 2016c). If significant human activity occurs inside a WHS it could potentially damage the ecological condition of that site and compromise its Outstanding Universal Value, and is therefore incompatible with the objectives of the World Heritage Convention (UNESCO 2015). If a site's condition and values are compromised it could be placed on the list of World Heritage in Danger and, ultimately, its World Heritage Status can be revoked if the ecological condition inside a site continues to decline to the extent it loses the values that are the basis for its listing. The consequences for a host nation could be substantial, since they would be denied access to the World Heritage Fund and other financial mechanisms, technical support provided by UNESCO and the Advisory Bodies, and lose the sustainable development opportunities a World Heritage Site creates (Conradin et al. 2014). Accurate and transparent monitoring and reporting of both the human pressures facing WHS, and the ecological condition within WHS is therefore essential for both host nations and UNESCO.

Current monitoring of WHS is summarised in site-level reports and surveys. This includes periodic reporting on progress and condition by States Parties on a 6-year regional cycle, reactive monitoring led by UNESCO and the Advisory Bodies in response to current issues, and site-level monitoring and evaluation systems (Hockings et al. 2006, Hockings et al. 2008, Stolton et al. 2012). The IUCN's World Heritage Outlook initiative and its expert-driven evaluations also provide important information on the conservation outlook for all WHS (Osipova et al. 2014). These monitoring approaches are important and capture diverse site-level data, but do not include monitoring based on globally comparable quantitative datasets. We argue that these current monitoring approaches could be further strengthened by additionally using globally comparable datasets to assess increases in human pressure or changes in ecological state such as forest loss (Leverington et al. 2010). Thanks to recent advances in remote sensing technology, globally comparable data on human pressure and ecological state is now available, allowing trends to be analysed across the entire network of WHS for the first time. This important baseline information allows States Parties to assess their progress in preserving their WHS and enables rapid reporting of their progress to the World Heritage Committee.

In this study we quantify changes in spatial and temporal patterns of human pressure and ecological state across the entire global network of WHS and their surrounding landscapes for the first time. We examine human pressure in WHS in 1993 and 2009 using the most comprehensive cumulative threat map available, the recently updated Human Footprint (Venter et al. 2016a, c) which is a temporally explicit map of eight anthropogenic pressures on the terrestrial environment. An increasingly popular approach for monitoring ecological state is to monitor forest cover, which responds to anthropogenic pressures (Nagendra et al. 2013, Tracewski et al. 2016). Therefore we also examine patterns of forest cover loss in WHS between 2000 and 2012 using high resolution maps of global forest cover (Hansen et al. 2013b). We identify which WHS have suffered the greatest forest loss, and largest increases in human pressure, as well as sites which are performing well at limiting these negative changes and maintaining their ecological integrity.

Methods

World Heritage Site Data

Data on WHS location, boundary and year of inscription was obtained from the 2015 World Database on Protected Areas (IUCN and UNEP-WCMC 2018). We applied filtering criteria to identify which WHS qualified for our analysis. Out of all natural sites, sites inscribed only under criterion (viii), which covers sites of geological importance including fossil sites and caves (UNESCO 1972), were excluded from this analysis, with the exception of *Vredefort Dome* in South Africa, *Phong Nha-Ke Bhang National Park* in Vietnam, *Lena Pillars Nature Park* in Russia *and Ischigualasto/Talampaya Natural Parks* in Argentina, because they are part of larger conservation areas. In addition, we constrained our analysis to terrestrial WHS, and the terrestrial component of marine WHS. Due to the 1km^2 resolution of the Human Footprint data, we chose to exclude WHS smaller than 5km^2. Initially 190 WHS qualified for our analysis.

Analyzing Human Pressure

To measure human pressure on the natural environment we used the recently updated Human Footprint (Venter et al. 2016a, c), which is a globally-standardised measure of cumulative human pressure on the terrestrial environment. The updated Human Footprint is based on the original methodology developed by Sanderson et al. (2002); however, the update is temporally explicit, quantifying changes in human pressure over the period 1993 to 2009. At a 1km^2 resolution, the Human Footprint includes global data on: built environments, crop lands, pasture lands, population density, night lights, railways, major roadways and navigable waterways. This makes the Human Footprint the most comprehensive cumulative threat map available (McGowan 2016). Still, it is important to note that it does not include data on all the possible threats and pressures facing WHS. Other threats, including invasive species (Bradshaw et al. 2007a), overabundant species (Ndoro et al. 2015), wildlife poaching (Plumptre et al. 2007, Wittemyer et al. 2014), tourism pressure (Li et al. 2008), and rapid climate change (Scheffer et al. 2015), are not directly accounted for in the Human Footprint data. Although in some cases the included pressure data, including population density, night lights, railways, major roadways and navigable waterways, can contribute to these threats (e.g. invasive species and some forms of poaching), we acknowledge that some threats are not well covered, which makes this a conservative assessment of threats.

In the Human Footprint, individual pressures were placed within a 0 - 10 scale and summed, giving a cumulative score of human pressure ranging from 0 - 50. A Human Footprint score below 3 indicates land which is predominantly free of permanent infrastructure, but may hold sparse human populations. A Human Footprint score of 4 is equal to pasture lands, and is a reasonable threshold of when land can be considered “human dominated” and species are likely to be threatened by habitat conversion (Watson et al. 2016b). A Human Footprint score of 7 is equal to agriculture, above which a landscape will contain multiple pressures, for example agriculture with roads and other associated infrastructure, and is therefore highly modified by humans.

To compare mean changes in Human Footprint between WHS and their surroundings, we calculated the mean change in Human Footprint between 1993 and 2009 in WHS and a surrounding 10 km buffer zone. Calculating the Human Footprint in surrounding buffer zones allows us to infer how much pressure a WHS is under from developments surrounding the protected area. Buffer zones were defined as a 10km buffer of land directly adjacent to and surrounding each WHS, and were created using the Geographic Information System ArcMap version 10.2.1. Because WHS inscribed post 1993 could potentially have been impacted before their inscription as a WHS, we included only sites inscribed during or before 1993 when calculating the change in Human Footprint (n = 94).

Analysing Forest Loss

To assess forest loss, we followed Hansen et al. (2013b), and defined forest cover as vegetation taller than 5m and forest loss as the complete removal of tree canopy at a 30m resolution (Hansen et al. 2013b). Hansen forest-cover change data was extracted and processed in the Google Earth Engine (http://earthengine.google.org/), a cloud platform for earth-observation data analysis. Sites which had zero percent forest cover in 2000 were excluded from the analysis. Only WHS inscribed during or before 2000 were included in the forest loss analysis (n = 134), since WHS inscribed post 2000 could potentially have been impacted before inscription. We then calculated total forest loss between the years 2000 and 2012 as a percentage of forest extent in 2000 for all WHS and buffer zones. We adapted JavaScript code developed by Tracewski et al. (2016) for analysing Hansen forest-cover data within specified spatial zones, which is freely available online (https://github.com/RSPB/IBA). Gain in forest cover was not included in this analysis for two reasons: young forests are unlikely to support forest-dependant species, and much of the gain can be attributed to monoculture plantations of oil palm or rubber which are major threats to tropical forests (Tropek et al. 2014). There are limitations of satellite-derived estimates of global forest change, such as an inability to differentiate between ecologically valuable forest and agro-forests, such as oil palm, and lower accuracy in more arid environments (Hansen et al. 2013b, Achard et al. 2014, Tropek et al. 2014). Likewise, ground truthing is required to infer the causes of forest loss since the dataset does not differentiate between ecologically harmful clearing, and purposeful clearing for example of invasive species, which has a conservation benefit. But even with these limitations, the Hansen et al. (2013b) forest data product is considered the most accurate global representation of temporal loss of forest available (McRoberts et al. 2016).

Results

Human Pressure

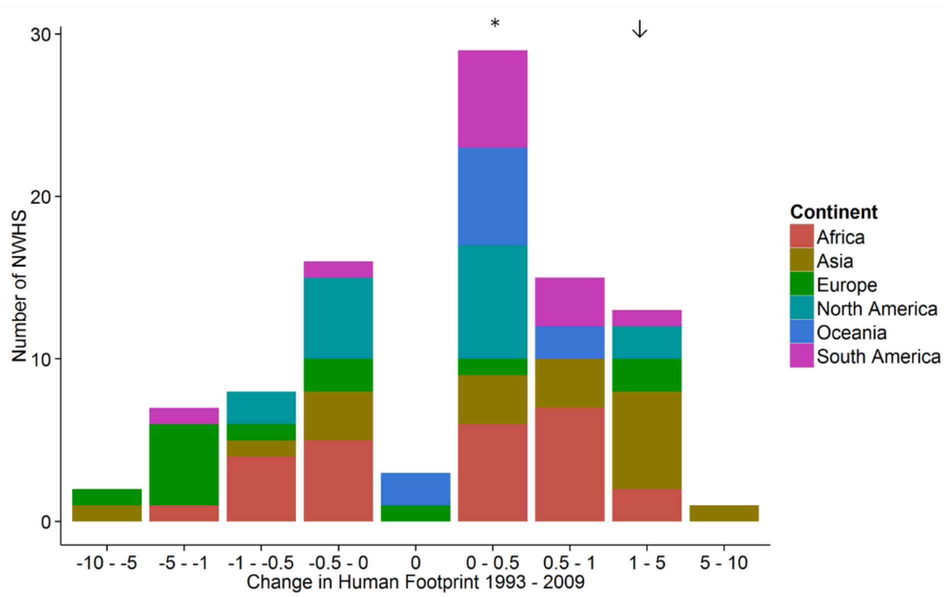
Human Pressure in WHS

The average Human Footprint per WHS in 2009 is 6.4, which is higher than the global average Human Footprint of 5.6, and there was considerable variation between regions and individual sites. Out of 94 WHS considered in this analysis, the majority of them (63%, n=59) had an average Human Footprint ≥ 4, and many WHS (38%, n=36) had a Human Footprint ≥ 7 meaning they are highly modified by humans. *Keoladeo National Park* in India was subject to the highest levels of human pressure of any WHS, with a 2009 Human Footprint of 23. *Göreme National Park* in Turkey, *Mount Taishan* in China, and *Manas Wildlife Sanctuary* in India were also subject to some of the highest levels of human pressure, with a Human Footprint of 19, 17 and 17 respectively. European and Asian WHS were under the highest levels of human pressure of all the continents, whereas WHS in North America and Oceania are under the lowest (Table 4.1). *Nahanni National Park* in Canada had the lowest 2009 Human Footprint of 0.08, along with *Kluane/ Wrangel-St. Elias/ Glacier Bay/ Tatshenshini-Alsek* in Canada/USA (0.3) and *Aïr and Ténéré Natural Reserves* in Niger (0.4). These three WHS are essentially free of human pressure but no WHS had a Human Footprint of zero (see supplementary Table 4.1 for a full list of WHS and their Human Footprint scores).

Table 4.1 Global and continental mean Human Footprint score per Natural World Heritage Site (WHS) and percentage change 1993 - 2009. Scores exceeding the global mean are shown in bold**.**

	Human Footprint 1993		Human Footprint 2009		% Change 1993 - 2009		
Continent	WHS	Buffer	WHS	Buffer	WHS	Buffer	# sites
Africa	6.0	6.9	6.2	7.1	**2.9**	2.8	25
Asia	**9.3**	**11.4**	**10.0**	**12.0**	**8.1**	**4.6**	18
Australia	3.3	4.2	3.6	4.6	**6.8**	**10.5**	10
Europe	**11.2**	**12.5**	**10.2**	**12.4**	-9.6	0.0	13
North America	2.8	3.9	2.9	4.0	**2.9**	2.6	16
South America	4.2	5.4	4.5	6.3	**4.8**	**15.8**	12
Global	6.3	7.4	6.4	7.8	1.7	4.5	94

Figure 4.1 Frequency distribution of changes in Human Footprint between 1993 and 2009 in Natural World Heritage Sites (WHS). * indicates the median change in HF and the arrow indicates the mean change in HF. Colors specify the continent in which the WHS is situated.



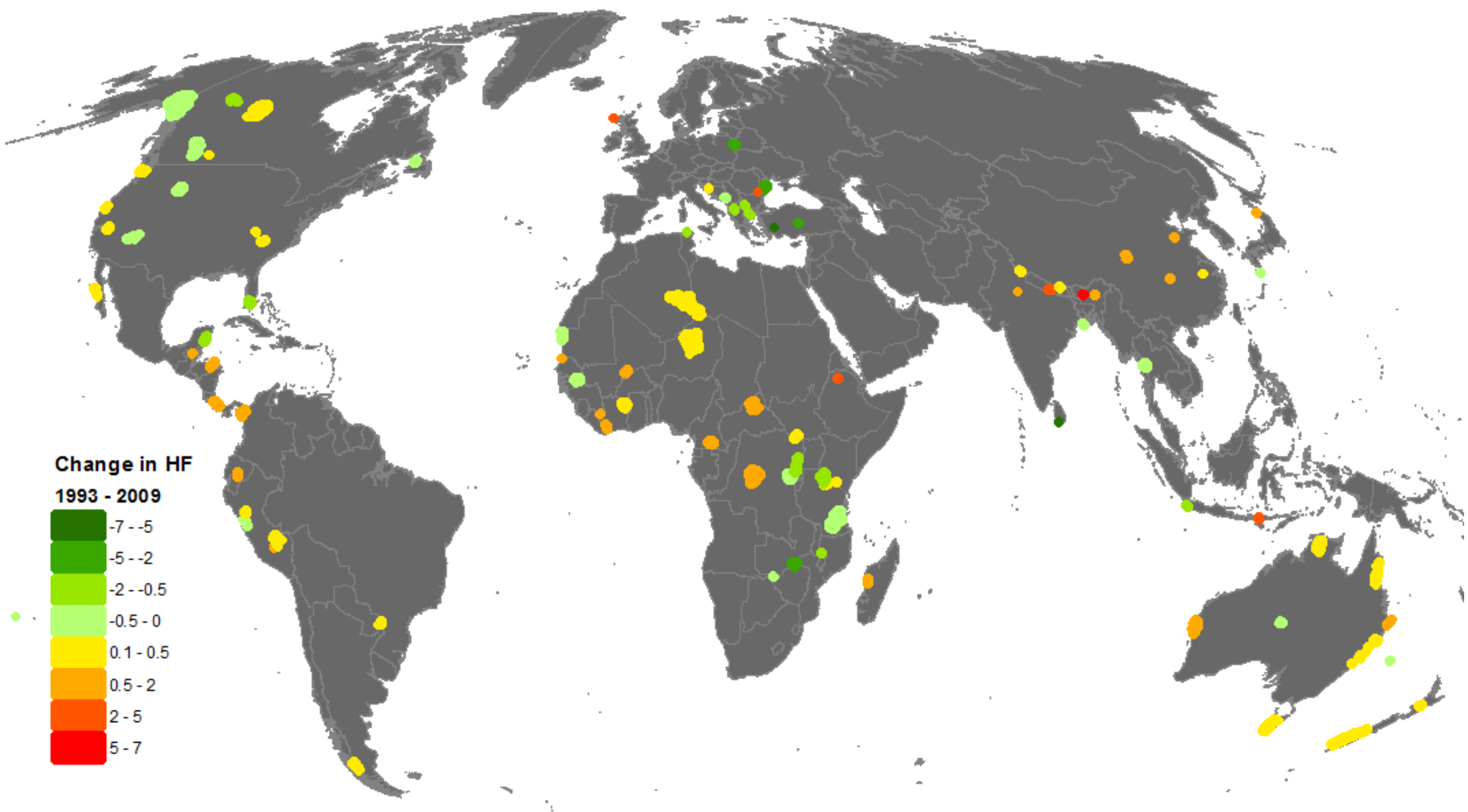
Changes in Human Pressure in WHS over time

The Human Footprint in WHS increased far more slowly than the global average, rising 1.7% between 1993 and 2009, compared to the global increase of 9%. However, human pressure did increase in the majority of WHS (63% n = 58) and across all continents except Europe (Figure 4.1). In most cases the increases were small; however, 14 sites (15%) were subject to substantial increases in human pressure (average Human Footprint increase > 1) (Table 4.2). The *Manas Wildlife Sanctuary* in India underwent the largest increase in human pressure of any WHS, with its Human Footprint rising by 5 to a score of 17 and is now one of the most highly modified by humans. *Komodo National Park* in Indonesia also underwent one of the largest increases in human pressure with its Human Footprint rising by 4.

Table 4.2 Natural World Heritage Sites (WHS) with the greatest increases and decreases in Human Footprint between 1993 and 2009.

	Human Footprint 1993		Human Footprint 2009		Change 1993 - 2009	
	WHS	Buffer	WHS	Buffer	WHS	Buffer
Increases						
Manas Wildlife Sanctuary	11.8	12.0	17.0	14.2	5.3	2.2
Komodo National Park	6.2	n/a	10.6	n/a	4.3	n/a
St Kilda	4.9	n/a	8.4	n/a	3.5	n/a
Chitwan National Park	11.5	13.9	14.5	17.5	3.0	3.5
Simien National Park	5.7	8.2	8.6	10.1	2.9	2.2
Decreases						
Sinharaja Forest Reserve	16.7	17.7	9.7	11.5	-7.0	-6.3
Hierapolis-Pamukkale	23.5	14.6	17.0	14.3	-6.5	-0.2
Bialowieża Forest	12.6	9.7	8.5	10.8	-4.1	1.2
Göreme National Park and the Rock Sites of Cappadocia	22.0	13.2	18.8	12.9	-3.3	0.0
Mana Pools National Park, Sapi and Chewore Safari Areas	9.0	8.9	6.2	6.7	-2.9	-2.2

Figure 4.2 Change in mean Human Footprint between 1993 and 2009 across Natural World Heritage Sites (WHS) inscribed prior to 1993. WHS which experienced an increase (which may threaten their unique values) are shown in red, whilst WHS which experienced a decrease are shown in green. Site boundaries are not to scale, and have been enlarged for clarity.



The largest increases in human pressure occurred in Asian WHS, where the regional mean Human Footprint increased by 8% between 1993 and 2009 (Figure 4.2). WHS in Oceania and South America also underwent relatively large increases in human pressure, with their mean Human Footprints rising by 6.8% and 4.3% respectively. The Human Footprint in European WHS decreased by 10% during the time period, however they were highly modified WHS to begin with and thus still face the highest levels of human pressure of all continents. Some notable decreases occurred in the *Sinharaja Forest Reserve* in Sri Lanka, *Hierapolis-Pamukkale* and *Göreme National Park* in Turkey, whose Human Footprint decreased by 7, 6.5 and 4 respectively.

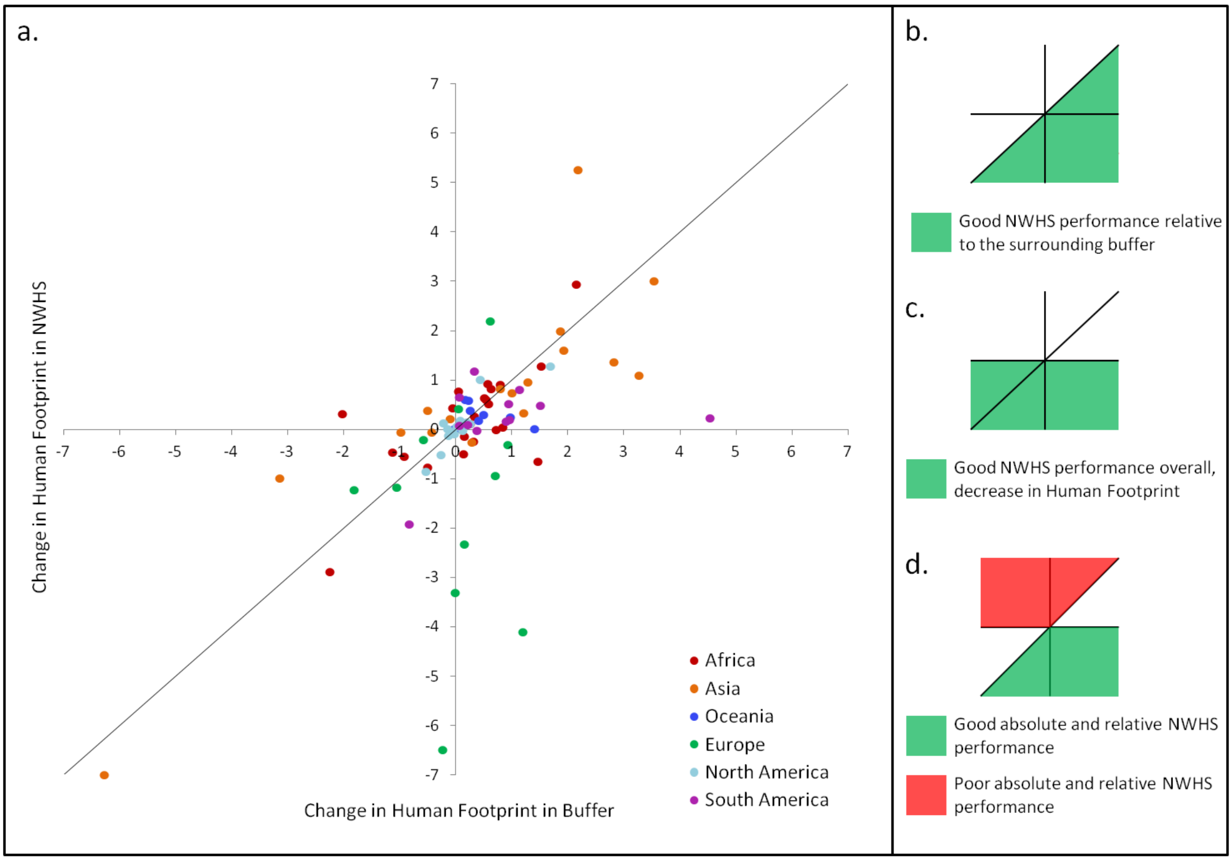
Comparison with Buffer Zones

The 2009 average Human Footprint per buffer zone is 7.8, which is slightly higher than the average Human Footprint per WHS of 6.4. The trend of human pressure being higher in the landscapes surrounding WHS held across all continents and for the majority of WHS (78% n=70). European and Asian WHS had the greatest levels of human pressure in their buffer zones, which were considerably higher than the global average. The *Danube Delta* in Romania had the greatest difference in human pressure compared to its buffer zone, with the relatively low 2009 average Human Footprint of 4.5 inside the WHS compared to a relatively high 13.9 in its buffer zone. Interestingly, some WHS such as *Sagarmatha National Park* in Nepal had very high levels of human pressure inside their boundaries compared to their buffer zones, with 2009 average Human Footprint scores of 6.5 and 3.7 respectively.

Globally, the average Human Footprint in buffer zones increased much faster than inside WHS, rising by 4.5% compared to 1.7% between 1993 and 2009 (Figure 4.3). These increases were largest in buffer zones in South America and Australia where the Human Footprint increased by 16% and 11% respectively. Many WHS performed well at limiting increases in human pressure relative to the amount of pressure they are under from the surrounding landscape. For example in *Iguaçu National Park* in Brazil the Human Footprint stayed almost constant within the WHS between 1993 and 2009, increasing by 0.2 compared to a large increase of 4.5 in its buffer zone. Likewise in *Mount Taishan* in China the Human Footprint only increased by 1.1 inside the WHS but by 3.3 in its buffer zone. Conversely, some WHS underwent larger increases in human pressure within their borders than in their buffer zones. These include *Manas Wildlife Sanctuary* in India where the Human Footprint inside the WHS increased by 5.3 compared to 2.2 in the buffer zone, and *Simien*

National Park in Ethiopia where the Human Footprint inside the WHS increased by 2.9, compared to 2.2 in its buffer zone.

Figure 5.3 (a) Change in Human Footprint between 1993 and 2009 inside Natural World Heritage Sites (WHS) versus buffer zones. WHS are coloured according to continent. (b) WHS below the identity line have undergone less change than their surrounding buffers indicating good relative performance. (c) WHS below the x-axis have undergone a mean decrease in Human Footprint indicating good overall performance. (d) We can visualise sites performing well on both the absolute and relative scales (green), or poorly on both (red).



Forest Cover Loss

Forest Loss in WHS

Forest loss occurred in the majority of forested WHS (91%, n=122) with a mean percentage loss of 1.48% per WHS (Figure 4.4). In the year 2000 there was 433,173 km^2 of forest cover inside all WHS and by the end of 2012 the total area of forest cover lost was 7,271 km^2 (1.67%). The majority of WHS suffered low levels of forest loss, with 72% (n=97) of WHS losing < 1%. However, 8% (n=11) of WHS suffered substantial forest loss (>5%), the majority of which are North American WHS (Figure 4.5). North American WHS accounted for 57% of all the forest lost in WHS globally (Table 4.3). *Waterton Glacier International Peace Park* that crosses the Canadian and USA border lost almost one quarter of its forested area (23%, 540km^2), *Wood Buffalo National Park* in Canada lost 12% (2,582km^2) of forest cover, and *Yellowstone National Park* in the USA lost 6% (217km^2)(Table 4.4). *Río Plátano Biosphere Reserve* in Honduras and *Lake Baikal* in Russia also lost large proportions of forest cover, 8% (365km^2) and 5% (1332km^2) respectively (see supplementary Table 4.2) for a full list of WHS and forest loss statistics). After North America, Asian and South American WHS lost the largest areas of forest within their WHS. WHS in Oceania lost an above average percentage of their forested area.

Figure 4.4 Frequency distribution of percent forest loss between 2000 and 2012 in Natural World Heritage Sites (WHS). * indicates the median % loss and the arrow indicates the mean % loss. Colours specify the continent in which the WHS is situated.

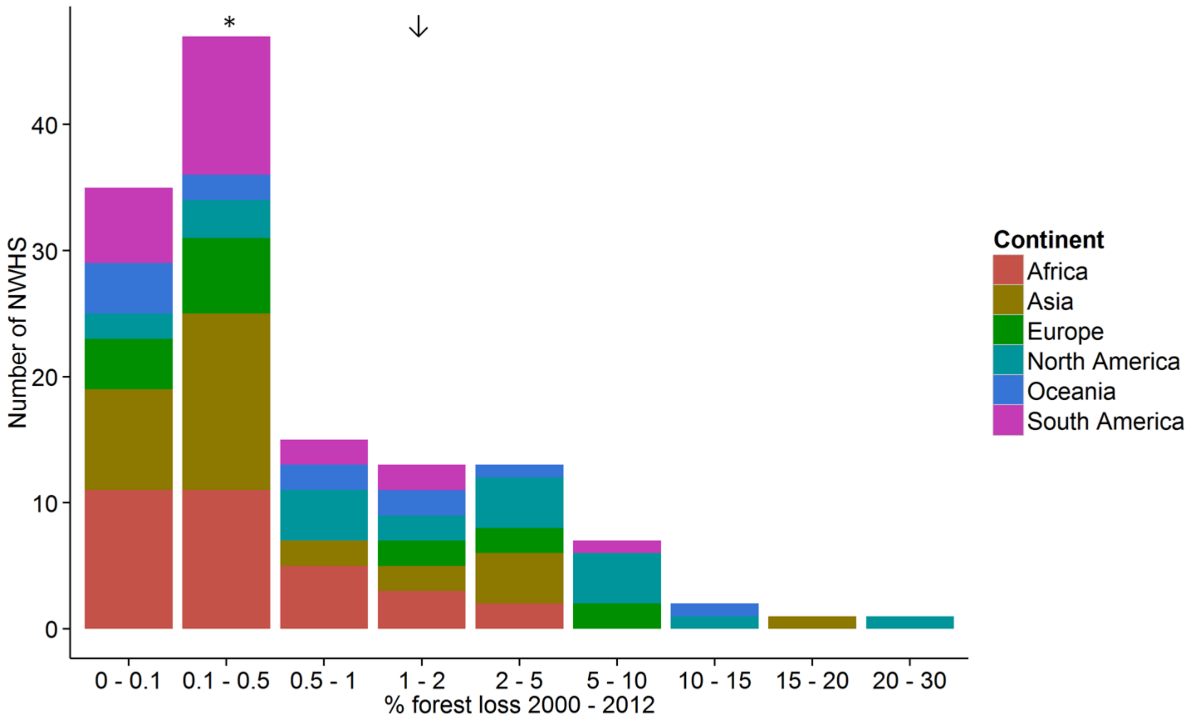
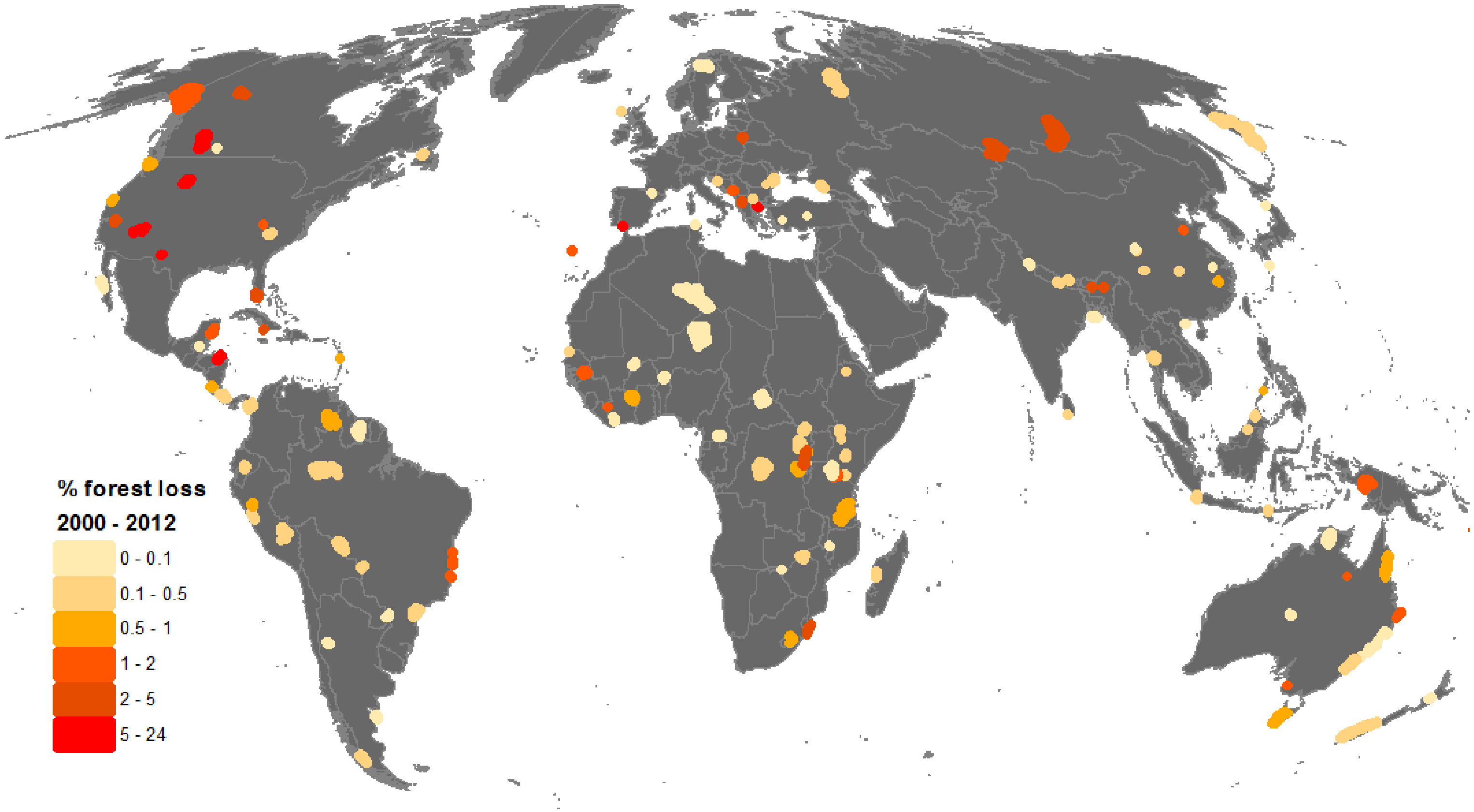


Figure 4.5 Percent forest loss between 2000 and 2012 in Natural World Heritage Sites inscribed prior to 2000. Sites experiencing substantial forest loss (>5%) are shown in red. Site boundaries are not to scale, and have been enlarged for clarity.



Forest Loss in Buffer Zones

Forest loss was higher in the buffer zones surrounding WHS than in the sites themselves with a mean percentage loss of 2.9% per WHS buffer zone. This trend held for all continents except for North America, where forest loss in the buffer zones was at very similar levels to inside WHS. WHS in Oceania lost the highest percentage of forest cover in their buffer zones and European WHS the least. There was a clear increase in the number of WHS suffering substantial forest losses of > 5% in their buffer zones (19% n=25), compared to within their boundaries. Forest loss was low (<1%) in only half of the WHS buffer zones (48% n=58), while 72% of WHS (n = 97) had low rates within their borders. Some notable WHS which lost large proportions of forest in their buffer zones are the *Australian Fossil Mammal Sites (Riversleigh / Naracoorte)* which lost 33% (9km^2), *The Discovery Coast Atlantic Forests* in Brazil which lost 11% (192 km^2), and *Kinabalu Park* in Malaysia which lost 10% (150 km^2). Many WHS performed well at limiting forest loss within their borders, despite considerable losses in their buffer zones (Figure 4.6). *Mount Wuyi* in China, for example lost only 1% (7km^2) within its borders compared to 9% (122 km^2) in its buffer zone. And *Iguazu National Park* in Argentina lost almost no forest inside its borders (0.02% <1km^2) compared to extensive loss in its buffer zone (13% 110km^2).

Table 4.3 Global and continental mean percentage forest loss per Natural World Heritage Site (WHS), and total area of forest lost between 2001 and 2012. Percentages exceeding the global average are shown in bold.

	Mean % forest loss per WHS		Summed forest loss (km2)		
Continent	WHS	Buffer	WHS	Buffer	# sites
Africa	0.6	2.4	523.4	1220.4	32
Asia	1.2	2.3	1599.2	1628.9	31
Australia	**1.6**	**6.2**	237.8	524.6	12
Europe	1.5	1.9	51.1	89.0	16
North America	**3.9**	**3.8**	4131.8	1814.3	21
South America	0.7	2.7	728.0	1479.3	22
Global	1.5	2.9	7271.2	6756.6	134

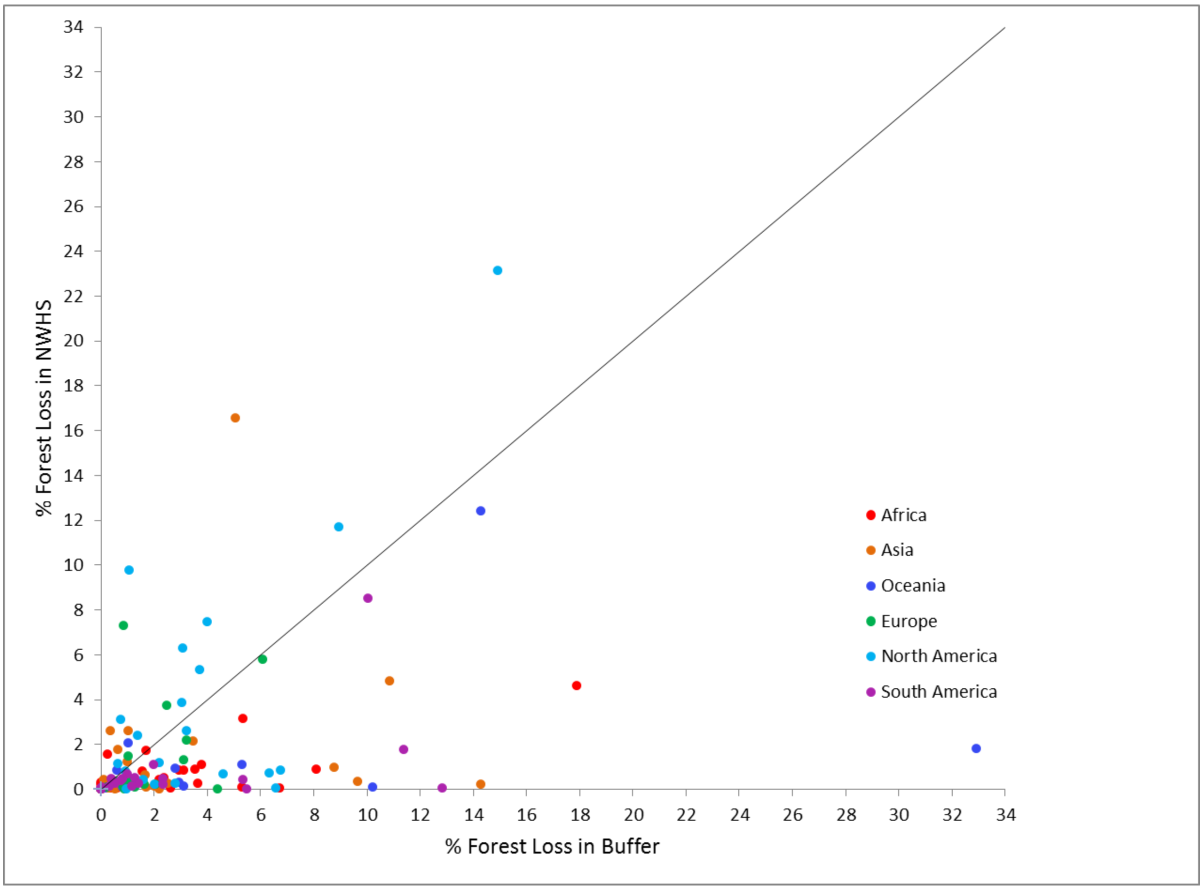
Table 4.4 Natural World Heritage Sites (WHS) with high percentage forest loss between 2001 and 2012. The total area of forest lost over the time period is also shown.

	% forest loss		Summed forest loss (km2)	
	WHS	Buffer	WHS	Buffer
Waterton Glacier International Peace Park	23.1	14.9	540.7	317.1
Shark Bay	12.4	14.3	5.8	2.7
Wood Buffalo National Park	11.7	8.9	2581.5	513.4
Grand Canyon National Park	9.8	1.1	38.2	5.1
Río Plátano Biosphere Reserve	8.5	10.1	365.6	252.0
Doñana National Park	7.3	0.8	2.1	1.0
Yellowstone National Park	6.3	3.1	217.0	59.4
Mount Athos	5.8	6.1	13.1	0.7
Canadian Rocky Mountain Parks	5.3	3.7	424.5	176.4
Lake Baikal	4.8	10.9	1332.6	1044.7

Discussion

Our analysis is the first globally comparable quantitative assessment of changes in human pressure and ecological state across the entire network of WHS, which is important baseline information for the UNESCO World Heritage Convention, the IUCN as the advisory body to UNESCO for Natural World Heritage, and the States Parties to monitor their progress at conserving WHS. We found that human pressure is increasing and forest loss is occurring in the majority of forested WHS worldwide, threatening to undermine their Outstanding Universal Value. Our most concerning finding is that a number of WHS are severely threatened by large increases in human footprint (>1) (14 WHS = 15% of the 94 WHS analyzed), and extensive forest loss (>5%) (11 WHS = 8% of the 134 WHS analyzed). The negative impact occurring in these sites requires large scale conservation interventions to ensure their value remains protected and sustained in the future. Our findings support qualitative assessments from case-by-case reports, which corroborates that WHS are becoming increasingly threatened globally, and that the condition of a third of WHS is now of significant concern (Osipova et al. 2014, Wang et al. 2014). Our results also support other studies showing that habitat extent and condition are declining in many protected areas across the globe (Laurance et al. 2012, Geldmann et al. 2014). However, our findings are particularly concerning since WHS are flagship protected areas afforded the highest level of international protection.

Figure 4.6 Percent forest loss between 2000 and 2012 in Natural World Heritage Sites (WHS) versus buffer zones. WHS are coloured according to continent. WHS below the identity line have suffered higher forest loss in the buffer zone compared to within the WHS boundaries.



There have been alarming rates of forest loss in the buffer zones surrounding nationally designated protected areas over the last three decades (DeFries et al. 2005, Bailey et al. 2016, Lui and Coomes 2016), and our results confirm this is also the case for many WHS. We found that forest loss and increases in human pressure were considerably higher in the buffer zones surrounding the vast majority of WHS. This suggests that WHS may be performing well at limiting negative changes within their boundaries (Bruner 2001). However, our findings clearly show that WHS are becoming increasingly isolated which is concerning since the ecological integrity of many WHS depend on links with the broader landscape (Naughton-Treves et al. 2005, Kormos et al. 2016). Environmental degradation around WHS could decrease their area and increase edge effects, which are important determinants of biodiversity persistence (Woodroffe and Ginsberg 1998, Hansen and DeFries 2007, Newmark 2008). Furthermore, Laurance et al. (2012) found that degradation occurring around a protected area strongly predisposes it to similar degradation within its borders, including trends in forest loss and human pressure. To avert further damage to WHS the World Heritage Committee should consider directing more resources to conservation in the landscapes surrounding WHS, and continue designating and strengthening official buffer zones around WHS, where communities are engaged and low impact land uses promoted (Laurance et al. 2012, UNESCO 2015, Kormos et al. 2016, Weisse and Naughton-Treves 2016).

We found that one third of WHS underwent a decrease in human pressure, which is a good result for conservation and a benchmark for other WHS and protected areas to strive towards. The Human Footprint decreased on average across European WHS, which is also encouraging, however we suggest that decreases in the Human Footprint should be interpreted with care. Although the Human Footprint is the most comprehensive cumulative threat map available, it does not include data on all the possible threats and pressures facing WHS, suggesting our results are conservative, and that WHS may be even more threatened than we have demonstrated. For example in *Aïr and Ténéré National Park* in Niger we found that changes in the Human Footprint were minimal (0.1) but understand that political instability and civil strife, along with poaching are the main pressures threatening the park (UNESCO 2016d). These limitations can be largely overcome by combining our data with site level case-by-case reports and therefore our study complements statutory monitoring mechanisms under the World Heritage Convention (UNESCO 2015) and IUCN's World Heritage Outlook initiative (Osipova et al. 2014). As discussed in the methods section, there are also limitations with satellite derived estimates of global forest change, for example it is impossible to infer the causes of forest loss without the use of site-level data, and not all forest loss in WHS is necessarily negative. For example, *iSimangaliso Wetland Park* in South Africa lost 18% (161km^2) of the forest in its buffer zone, but this is due to the purposeful clearing of pine and eucalyptus plantations for restoration (Zaloumis and Bond 2011), so clearly serves a positive conservation purpose. However, given the impacts of habitat loss on biodiversity (Maxwell et al. 2016) and the prevalence of forest loss in protected areas globally (Heino et al. 2015), we do assume in the majority of cases that forest loss is detrimental to the ecological state of WHS. We also note that forest loss is also just one indicator of ecological state, and a measure of intact forest cover does not necessarily guarantee a WHS is in good condition. For example the *Dja Faunal Reserve* in

Cameroon lost almost no forest during the time period; however, it has suffered intense poaching in recent times threatening wildlife populations within its borders (UNESCO 2016e). The limitations of remotely sensed data are widely recognized and need to be acknowledged, yet it remains an increasingly important tool for conservation monitoring, and its overall utility is broadly acknowledged (Turner et al. 2003, Buchanan et al. 2009, Tracewski et al. 2016).

Conclusion

The World Heritage Convention should be one of the world's most effective conservation instruments globally, identifying and protecting the Earth's most valuable natural landscapes. Our aim is to highlight growing challenges which are undermining its success. New globally comparable data sets such as the Human Footprint and the Global Forest Change data have provided an urgently needed opportunity to measure how well WHS are maintaining their ecological integrity (Watson et al. 2016a). We used these metrics to analyse spatial and temporal trends in human pressure for 94 WHS, and forest loss in 134 WHS, presenting baseline data for the World Heritage Committee and the States Parties. There is a clear opportunity for the World Heritage Committee to establish thresholds and targets with regard to human pressure and forest loss in WHS, and measure the effectiveness of management interventions across sites. We urge the World Heritage Committee to assess the status of the WHS which our analysis suggests are highly threatened, since urgent conservation intervention is now clearly needed to save many of these WHS and their outstanding and unique values in perpetuity.

CHAPTER 5 Gaps and opportunities for the World Heritage Convention to contribute to global wilderness conservation

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Abstract

Wilderness areas are ecologically intact landscapes predominantly free of human uses, especially industrial scale activities, which result in significant biophysical disturbance. This definition does not exclude indigenous peoples and local communities who live in wilderness areas, depending on them for subsistence, and who have developed deep bio-cultural connections. Wilderness areas are important for biodiversity conservation, along with sustaining key ecological processes, and ecosystem services that underpin planetary life-support systems. Despite these widely recognized benefits and values they are insufficiently protected and are consequently being rapidly eroded. There are increasing calls for multilateral environmental agreements to make a greater and more systematic contribution to wilderness conservation before it is too late. We developed updated global maps of terrestrial wilderness and assessed wilderness coverage by the World Heritage Convention, one of the most important international conservation instruments. We found that one quarter of Natural and Mixed World Heritage Sites (WHS) contain wilderness, conserving a total of 545,307 km^2 (approximately 1.8% of the world's wilderness extent). Many WHS had excellent wilderness coverage such as the *Okavango Delta* in Botswana (11,914 km^2) and the *Central Suriname Nature Reserve* in Suriname (16,029 km^2). However, 22 (35%) of the world's terrestrial biorealms do not have any wilderness representation within WHS. As an efficient means of filling these gaps, we identify 840 protected areas > 500 km^2 in size which are predominantly wilderness (>50% of their area) and represent 18 of these 22 missing biorealms. These offer a starting point for assessing the potential for the designation of new WHS that could help increase wilderness representation on the World Heritage List. We also urge the World Heritage Convention to help ensure that the ecological integrity and Outstanding Universal Value of existing World Heritage Sites with wilderness values is preserved.

Introduction

Wilderness areas are ecologically intact landscapes, predominantly free of disruption and degradation by large-scale human disturbances (Lesslie et al. 1988, Mackey et al. 1998, Watson et al. 2016c). They are not exclusive of people as many support indigenous peoples and local communities who have long inhabited these lands, are often politically and economically marginalised, and whose rights should be respected at all times (Gorenflo et al. 2012, Schwartzman et al. 2013). Instead, wilderness areas are free of large-scale land conversion, dense human settlements, industrial activity and infrastructure development (Watson et al. 2016c), which lead to significant biophysical disturbance of the natural environment (Mittermeier et al. 2003, Kormos et al. 2016).

Free from the disruptive impacts of direct human pressures, wilderness areas continue to support key evolutionary and ecological processes largely unimpeded (Klein et al. 2009, Watson et al. 2009, Martin and Watson 2016). These processes generate a range of high-value ecosystem services including regulation of hydrological cycles at multiple scales (Salati et al. 1979, Furniss et al. 2010) and significant organic carbon stocks (Mackey et al. 2013). Wilderness areas are the only reference places on Earth remaining where we can study how natural systems operate largely free from the direct impacts of modern industrial society, and as such, are a baseline reference and source of propagules and populations for restoration and re-wilding efforts (Watson et al. 2016c).

Wilderness areas are also critically important for *in situ* biodiversity conservation, supporting the last intact mega-faunal assemblages (Mittermeier et al. 2003, Ripple et al. 2015), wide-ranging and migratory species (Klein et al. 2009, Bauer and Hoye 2014), and species which are sensitive to exploitation or conflicts with humans (Ripple et al. 2014). As such species rapidly disappear from human dominated landscapes, wilderness areas are becoming their last remaining strongholds (Gibson et al. 2011). Similarly, human cultural and language diversity which co-occurs with biodiversity is also declining outside of wilderness areas (Gorenflo et al. 2012). The sustainable livelihoods and cultural integrity of many indigenous communities are often threatened by the same industrialized development pressures that threaten biodiversity (Boff 2002).

Despite the well documented environmental, ecological and bio-cultural values of wilderness areas, they have not been regarded as a conservation priority (Myers et al. 2000, Brooks et al. 2006), and there is still no explicit and systematic recognition of their importance in powerful multilateral environmental agreements such as the Convention on Biological Diversity or the World Heritage Convention. However, wilderness areas are increasingly under threat and have suffered catastrophic declines in extent over the last two decades, comprising one tenth (3.3 million km^2) of the world's wilderness area (Watson et al. 2016c). Human pressure has been documented as spreading into almost all remaining wilderness areas, reducing their extent, degrading their intactness, causing loss of biodiversity and undermining the resilience of human communities (Gorenflo et al. 2012, Laurance et al. 2015, Ibisch et al. 2016, Venter et al. 2016c, Potapov et al. 2017). It is also of concern that efforts to protect wilderness areas over the last two decades have failed to keep pace with the rate of wilderness loss (Watson et al. 2016c). There is clearly an immediate need for international and national policies to recognise the importance of conserving wilderness areas, raising their profile, communicating their irreplaceability, and promoting their protection. This is particularly important in countries where national policies or legislation are weak or not adequately implemented.

There have been recent and timely calls for the World Heritage Convention to recognize the significance of wilderness conservation (Kormos et al. 2016). The World Heritage Convention was adopted in 1972 to conserve the world's most valuable natural and cultural sites (UNESCO 1972). Since then, 193 governments have become States Parties to the World Heritage Convention thereby committing to conserve the 1,052 World Heritage Sites listed to date (UNESCO 2016f). These places are deemed to have "Outstanding Universal Value" (OUV), meaning they are so exceptional as to transcend national boundaries and are important for present and future generations of all humanity (UNESCO 2015). OUV is defined in the Convention's Operational Guidelines based on three pillars; a site must meet one of the ten criteria for listing, as well as demonstrating "integrity" and "intactness" of their values, and have adequate long-term official protection and management efforts in place (UNESCO 2015). The criteria for defining OUV of Natural World Heritage Sites, of which there are 203, are aesthetic value and natural phenomena (vii), geological value (viii), ecological and biological processes (ix), and biodiversity (x) (UNESCO 2015). There are also 35 Mixed World Heritage Sites designated for meeting at least one of the natural heritage criteria and one of the cultural heritage criteria (hereafter 'WHS' refers to both Natural and Mixed Sites which are the focus of this study).

World Heritage status cannot be granted purely because a place is a wilderness area, although high wilderness quality can be associated with all four of the current natural criteria, as well as the requirements for integrity and intactness (UNESCO 2015). The *Tasmanian Wilderness* in Australia is an example of a mixed WHS, meeting all four natural criteria plus three cultural criteria, where wilderness quality is a critical consideration. It has been argued that wilderness areas already make essential contributions to the OUV of many current WHS, and wilderness quality must be protected to ensure the integrity of these sites is maintained (Kormos et al. 2016). Wilderness areas can therefore be used to guide the identification of potential new sites, and wilderness quality should be fully considered when they are being assessed for OUV. Previous efforts to assess the World Heritage Convention's coverage of wilderness areas identified serious caveats from using outdated maps of wilderness areas (Bertzky et al. 2013, Kormos et al. 2016).

In this study, we created a new map of large terrestrial wilderness areas utilising recently updated maps of human pressure on the environment (Venter et al. 2016a, c). We used this map to assess the current coverage of wilderness areas by the World Heritage Convention, and to identify potential gaps in coverage. We then identified large nationally designated protected areas with good wilderness coverage within gaps, which could potentially be designated as new WHS if they meet the other conditions of integrity and OUV. The World Heritage List is still being expanded, presenting an important opportunity to take wilderness areas into account when assessing potential new WHS, and to make a greater contribution to their ongoing conservation by adding an important layer of protection.

Methods

To map the global extent of remaining large wilderness areas we used the methodological framework outlined in Sanderson et al. (2002), but utilised the recently updated 'Human Footprint' map of Venter and colleagues (2016a; 2016b). The latter is a globally-standardised map of cumulative human pressure on the terrestrial environment. At 1 km^2, it is the finest resolution cumulative threat map available, as well as the most comprehensive (McGowan 2016), including data on eight human pressures globally: built environments; crop lands; pasture lands; population density; night lights; railways; major roadways; and navigable waterways. These eight individual human pressures were standardised on a 0-10 scale based on their estimated contribution to human influence on the natural environment following Sanderson et al. (2002). The standardised scores were then summed, giving a total cumulative pressure score out of fifty for each pixel (some pressures are mutually exclusive, whilst others can co-occur).

It is important to note that the Human Footprint relies on datasets that are globally comparable, but in some areas may not have the full extent of infrastructure that national or sub-national datasets contain. It therefore sometimes maps pressures as absent where they are actually present, underestimating human pressure in those parts of the world. The Human Footprint also does not reflect all the pressures that could potentially impact on the wilderness quality of an area. For example, threats such as poaching, logging, invasive species, pollution and climate change are not directly captured, although many of them are often highly correlated to the pressures that were included in the analysis (Venter et al. 2016a, c).

To identify a set of wilderness areas that are of global significance, we first created a layer of 62 biorealms as a biogeographic framework for our analysis, based on the widely used Terrestrial Ecoregions of the World (Olson et al. 2001). The biorealms represent all existing combinations of the world's 14 vegetated biomes and seven biogeographic realms (for example boreal forests exist in both the Palearctic and Nearctic realms), following the established practice of excluding Antarctica and other rock and ice areas (Juffe-Bignoli et al. 2014, Venter et al. 2014b). We then identified the 10% area within each biorealm with the lowest Human Footprint score, following the same methods as Sanderson et al. (2002). From this, we selected all contiguous areas > 10,000 km^2. In cases where a biorealm did not contain at least ten contiguous patches > 10,000km^2, we consecutively selected the next largest patch until we had a total of ten patches per biorealm, or failing this, all patches per biorealm. These were used to generate our updated map of terrestrial wilderness areas (Figures 5.1 and 5.2).

Figure 5.1 The extent of terrestrial wilderness areas in 2009 and the current extent of Natural and Mixed World Heritage Sites. Wilderness areas with World Heritage coverage are shown in grey, and gaps in coverage are shown in green.

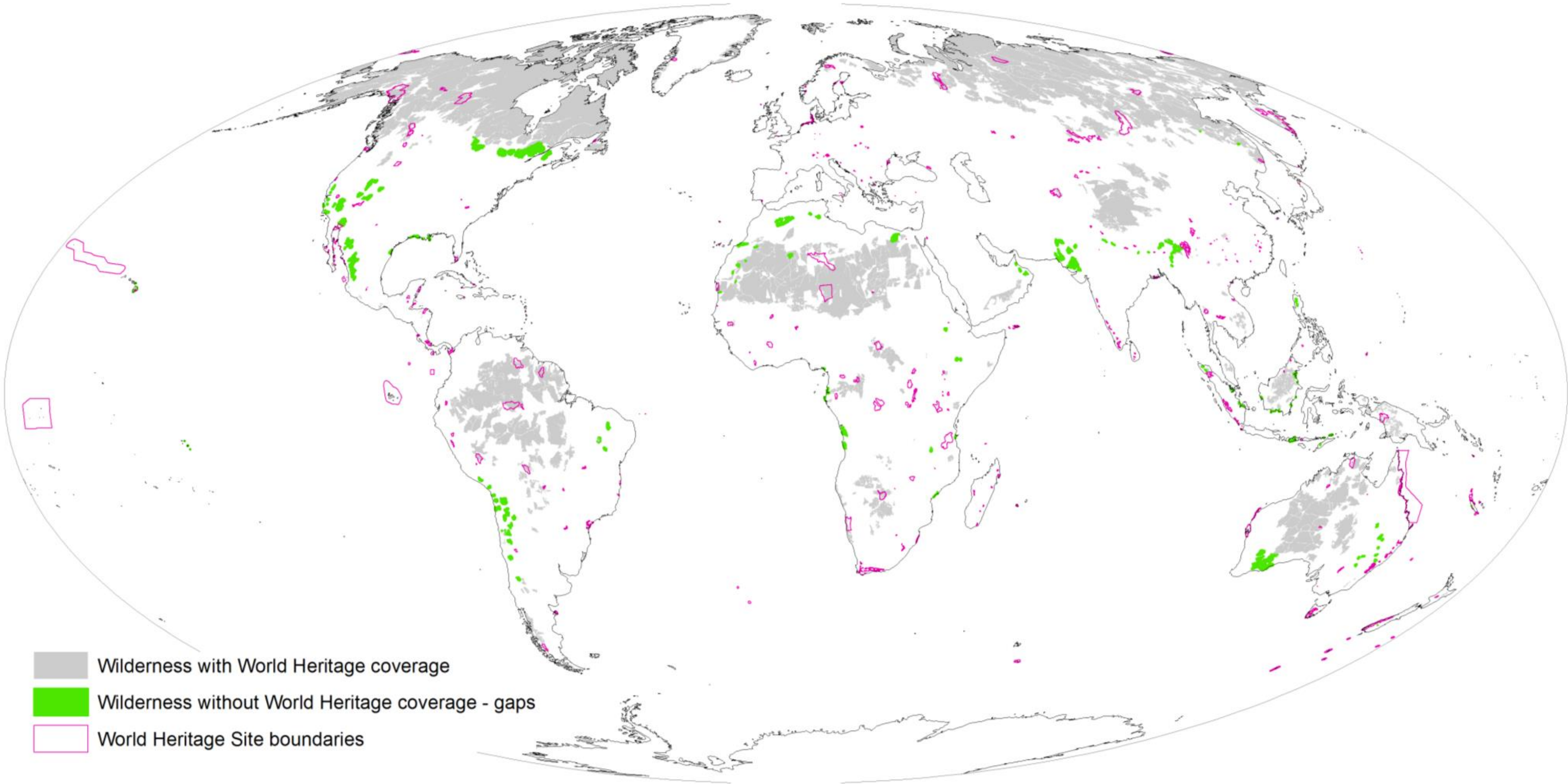
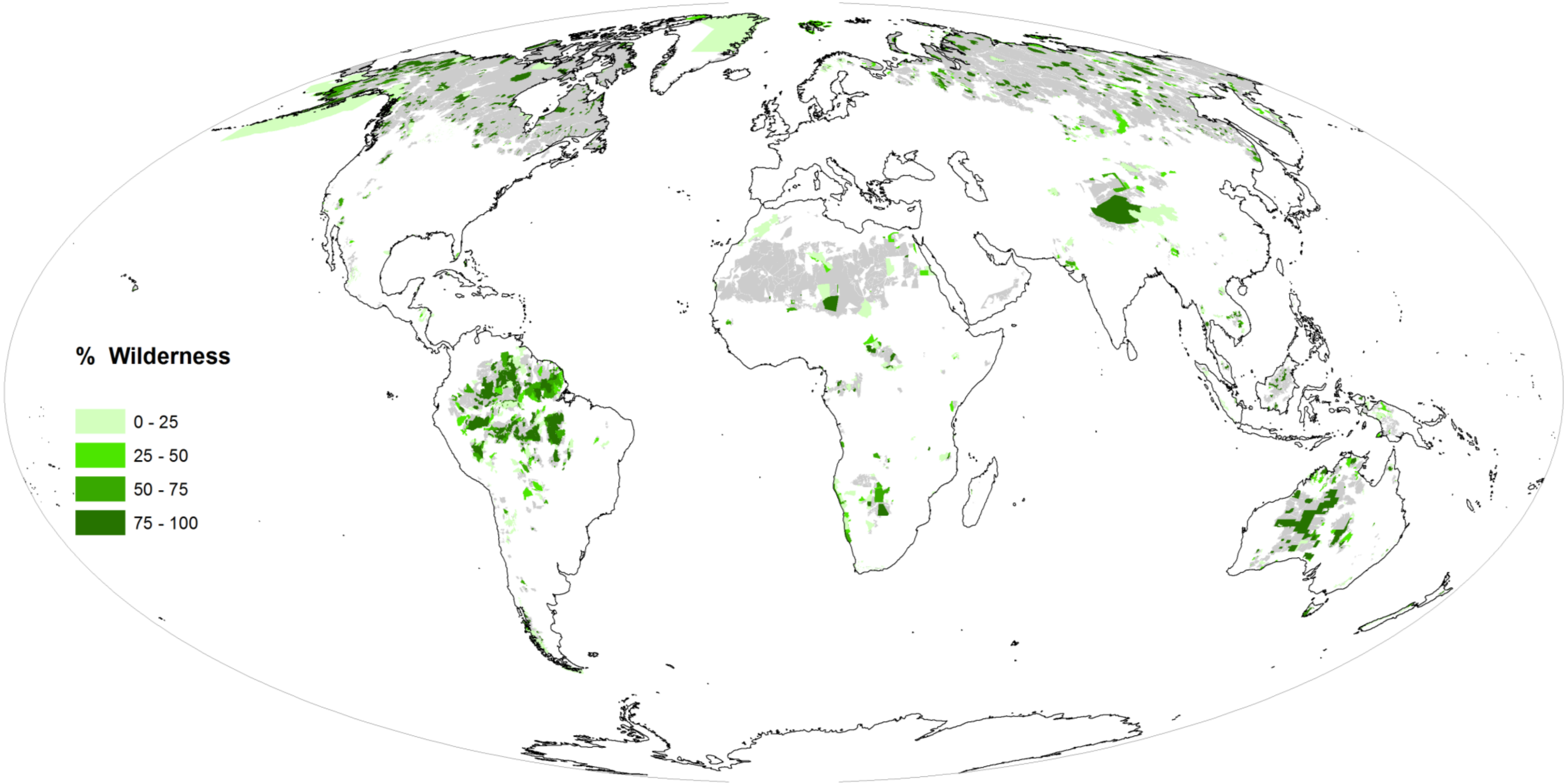


Figure 5.2 All protected areas larger than 500 km^2 with wilderness area coverage, and the percentage of their area which is wilderness. Remaining extent of wilderness is shown in grey.



We obtained data on protected areas and Natural and Mixed World Heritage Site locations and boundaries from the 2016 World Database on Protected Areas (IUCN and UNEP-WCMC 2018). We followed previous assessments (Jenkins et al. 2013, Butchart et al. 2015) and included only protected areas with a national designation. We excluded protected areas which only have central coordinates rather than a polygonal representation in the WDPA database. We calculated the current coverage of terrestrial wilderness within the 208 Natural and Mixed World Heritage Sites which have some terrestrial area, and were designated at the time of this study (May 2016). We also calculated the percentage of wilderness area protected in WHS within each biorealm and biome. Finally, we calculated the extent of wilderness area coverage within all protected areas > 500 km^2 (hereafter “large protected areas”) to identify possible candidate WHS with a high level of wilderness area coverage within the previously identified gaps. While there is no definitive scientific basis to this specific threshold, and it is well documented that smaller areas can retain some wilderness quality and ecologically important functions and values, it is also apparent that biological diversity and ecosystem resilience scales with geographic extent (Thompson et al. 2009) and that areas above this threshold will maintain many large scale ecological processes, including viable populations of space-demanding fauna (Soule et al. 2004a). The Convention’s Operational Guidelines emphasize the importance of WHS to be of adequate size to ensure the representation and long-term conservation of the features and processes that are of OUV (UNESCO 2015).

Results

Wilderness areas currently extend across almost a quarter of the world’s terrestrial area (22.7%) including all 14 biomes and 62 biorealms (Fig 5.1). Wilderness quality is not evenly distributed, with the largest extents occurring in the *boreal and taiga forests* (9,349,732km^2, 62% of biome extent), *tundra* (6,623,675km^2, 80%) and *desert and xeric shrublands* (6,470,715km^2, 23%), whilst *mangroves* and *tropical and subtropical coniferous forests* had the smallest estimated wilderness extents (22,661 km^2, 7% and 57,341 km^2, 8% respectively) (Table 5.1).

Out of the 208 WHS considered in this analysis, one quarter (25% n=52) contain wilderness areas as defined above (Table S5.1). Twelve WHS (6%) had a high level of coverage (>90% of WHS is within defined wilderness areas) including the *Putorana Plateau* in Russia, *Nahanni National Park* in Canada, *Central Suriname Nature Reserve* in Suriname, and

Purnululu National Park in Australia, whilst 25 WHS (12%) had good wilderness coverage (>50% wilderness). The *Okavango Delta* in Botswana alone accounts for 80% (11,914 km^2) of the *flooded grasslands and savannas* wilderness protected within WHS globally. Of the 25 WHS with >50% wilderness, 12 sites cover over 10,000 km^2 of wilderness each, another 12 sites have between 1,000 and 10,000 km^2 of wilderness, and only the 526 km^2 *Gunung Mulu National Park* in Malaysia has < 500 km^2 wilderness (which still constitutes 90% of the site area).

We found that WHS currently protect 545,307 km^2 of the identified wilderness areas, amounting to 1.8% of the world's total terrestrial wilderness area. This protection occurs across all 14 biomes with the greatest wilderness area coverage occurring in *flooded grasslands and savannas* (14.7% of wilderness area in this biome, 3 sites), *mangroves* (11%, 2 sites), and *tropical and subtropical dry broadleaf forests* (9%, 11 Sites) (Table 5.1). However, gaps are evident with <1% of wilderness protected in *tropical and subtropical coniferous forests*, and *temperate grasslands, savannas and shrublands* (1 and 3 sites respectively), and <2% of wilderness protected in seven biomes. Large gaps are also evident across biorealms, with wilderness in 35% (n=22) of realms not protected in WHS (Table 5.2, Fig 5.1).

Table 5.1 Wilderness in each biome (Olson 2001), and wilderness coverage by Natural and Mixed World Heritage Sites (WHS).

Biome Name	*Area of Biome (km2)*	*Area of Wilderness (km2)*	*Area of Wilderness in WHS (km2)*	*% of Biome Wilderness*	*% of Wilderness in WHS*	*# of WHS*
Flooded grasslands and savannas	1,096,130	101,545	14,889	9.3	14.7	3
Mangrove	348,519	22,661	2,522	6.5	11.1	2
Tropical and subtropical dry broadleaf forests	3,025,999	170,212	15,185	5.6	8.9	5
Temperate broadleaf and mixed forests	12,835,688	544,189	29,649	4.2	5.4	5
Tropical and subtropical grasslands savannas	20,295,424	1,656,151	53,384	8.2	3.2	9
Temperate coniferous forests	4,087,094	707,544	21,163	17.3	3.0	5
Tropical and subtropical moist broadleaf forests	19,894,149	3,628,627	95,425	18.2	2.6	11
Mediterranean forests woodlands and scrub	3,227,266	125,260	2,313	3.9	1.8	1
Montane grasslands and savannas	5,203,411	760,651	13,764	14.6	1.8	4
Tundra	8,311,584	6,623,675	107,290	79.7	1.6	6
Deserts and xeric shrublands	27,984,645	6,470,715	89,427	23.1	1.4	9
Boreal forests	15,077,946	9,349,732	99,254	62.0	1.1	9
Temperate grasslands savannas and shrublands	10,104,080	214,074	995	2.1	0.5	3
Tropical and subtropical coniferous forests	712,618	57,241	47	8.0	0.1	1

Table 5.2 Wilderness in each biorealm (Olson 2001), and wilderness coverage by Mixed and Natural World Heritage Sites (WHS). Gaps in coverage are shown in bold.

Realm Name	*Biome Name*	*Area of Biorealm (km2)*	*Area of Wilderness (km2)*	*Area of Wilderness in WHS (km2)*	*% of Biorealm Wilderness*	*% of Wilderness in WHS*
Afrotropic	Deserts and xeric shrublands	2,408,199	138,017	12,041	5.7	8.7
Afrotropic	Flooded grasslands and savannas	458,825	34,533	11,914	7.5	34.5
Afrotropic	Mangrove	76,883	5,544	-	7.2	**0.0**
Afrotropic	Mediterranean forests woodlands and scrub	95,862	5,525	2,313	5.8	41.9
Afrotropic	Montane grasslands and savannas	864,245	20,540	-	2.4	**0.0**
Afrotropic	Temperate grasslands savannas and shrublands	25,841	3,864	-	15.0	**0.0**
Afrotropic	Tropical and subtropical dry broadleaf forests	195,296	11,118	1,081	5.7	9.7
Afrotropic	Tropical and subtropical grasslands savannas	14,012,118	1,055,211	31,981	7.5	3.0
Afrotropic	Tropical and subtropical moist broadleaf forests	3,493,130	187,599	4,017	5.4	2.1
Australasia	Deserts and xeric shrublands	3,580,113	1,585,016	549	44.3	0.0
Australasia	Mangrove	26,885	4,491	1,367	16.7	30.4
Australasia	Mediterranean forests woodlands and scrub	805,436	76,363	-	9.5	**0.0**
Australasia	Montane grasslands and savannas	67,648	6,504	1,913	9.6	29.4
Australasia	Temperate broadleaf and mixed forests	736,811	43,613	22,870	5.9	52.4

Australasia	Temperate grasslands savannas and shrublands	631,023	9,784	-	1.6	**0.0**
Australasia	Tropical and subtropical dry broadleaf forests	88,348	5,226	-	5.9	**0.0**
Australasia	Tropical and subtropical grasslands savannas	2,170,610	359,227	8,651	16.5	2.4
Australasia	Tropical and subtropical moist broadleaf forests	1,160,343	162,659	7,575	14.0	4.7
Australasia	Tundra	876	*n/a	n/a	n/a	n/a
Indo-Malay	Deserts and xeric shrublands	1,089,109	44,647	-	4.1	**0.0**
Indo-Malay	Flooded grasslands and savannas	27,965	10,016	-	35.8	**0.0**
Indo-Malay	Mangrove	119,125	4,149	-	3.5	**0.0**
Indo-Malay	Montane grasslands and savannas	4,349	860	173	19.8	20.1
Indo-Malay	Temperate broadleaf and mixed forests	149,971	11,774	-	7.9	**0.0**
Indo-Malay	Temperate coniferous forests	67,304	6,460	-	9.6	**0.0**
Indo-Malay	Tropical and subtropical coniferous forests	95,956	5,620	-	5.9	**0.0**
Indo-Malay	Tropical and subtropical dry broadleaf forests	1,531,782	60,226	5,227	3.9	8.7
Indo-Malay	Tropical and subtropical grasslands savannas	34,657	892	-	2.6	**0.0**
Indo-Malay	Tropical and subtropical moist broadleaf forests	5,422,850	346,106	4,779	6.4	1.4
Nearctic	Boreal forests taiga	5,103,133	4,355,904	56,139	85.4	1.3
Nearctic	Deserts and xeric shrublands	2,324,734	68,588	16	3.0	0.0

Nearctic	Mediterranean forests woodlands and scrub	121,535	8,772	-	7.2	**0.0**
Nearctic	Temperate broadleaf and mixed forests	2,842,613	179,266	-	6.3	**0.0**
Nearctic	Temperate coniferous forests	2,306,570	443,358	11,619	19.2	2.6
Nearctic	Temperate grasslands savannas and shrublands	3,096,883	65,799	50	2.1	0.1
Nearctic	Tropical and subtropical coniferous forests	289,050	35,260	-	12.2	**0.0**
Nearctic	Tropical and subtropical dry broadleaf forests	51,096	3,413	-	6.7	**0.0**
Nearctic	Tropical and subtropical grasslands savannas	80,803	7,276	-	9.0	**0.0**
Nearctic	Tundra	4,253,628	3,717,951	67,487	87.4	1.8
Neotropic	Deserts and xeric shrublands	1,178,911	24,319	-	2.1	**0.0**
Neotropic	Flooded grasslands and savannas	270,986	31,910	2,975	11.8	9.3
Neotropic	Mangrove	125,626	8,477	1,155	6.7	13.6
Neotropic	Mediterranean forests woodlands and scrub	148,840	8,146	-	5.5	**0.0**
Neotropic	Montane grasslands and savannas	874,755	50,016	-	5.7	**0.0**
Neotropic	Temperate broadleaf and mixed forests	413,204	79,005	3,440	19.1	4.4
Neotropic	Temperate grasslands savannas and shrublands	1,629,667	41,020	205	2.5	0.5
Neotropic	Tropical and subtropical coniferous forests	327,612	16,360	47	5.0	0.3
Neotropic	Tropical and subtropical dry broadleaf forests	1,144,759	88,292	8,706	7.7	9.9

Neotropic	Tropical and subtropical grasslands savannas	3,993,839	232,551	12,559	5.8	5.4
Neotropic	Tropical and subtropical moist broadleaf forests	9,277,772	2,908,849	78,955	31.4	2.7
Oceania	Tropical and subtropical dry broadleaf forests	14,717	1,936	171	13.2	8.8
Oceania	Tropical and subtropical grasslands savannas	3,397	994	193	29.3	19.4
Oceania	Tropical and subtropical moist broadleaf forests	29,041	1,510	-	5.2	**0.0**
Palearctic	Boreal forests taiga	9,974,812	4,993,827	43,115	50.1	0.9
Palearctic	Deserts and xeric shrublands	17,403,579	4,610,128	76,821	26.5	1.7
Palearctic	Flooded grasslands and savannas	338,354	25,085	-	7.4	**0.0**
Palearctic	Mediterranean forests woodlands and scrub	2,055,593	26,454	-	1.3	**0.0**
Palearctic	Montane grasslands and savannas	3,392,415	682,730	11,678	20.1	1.7
Palearctic	Temperate broadleaf and mixed forests	8,693,090	230,530	3,339	2.7	1.4
Palearctic	Temperate coniferous forests	1,713,220	257,725	9,544	15.0	3.7
Palearctic	Temperate grasslands savannas and shrublands	4,720,666	93,608	740	2.0	0.8
Palearctic	Tropical and subtropical moist broadleaf forests	511,013	21,905	99	4.3	0.5
Palearctic	Tundra	4,057,079	2,905,724	39,803	71.6	1.4

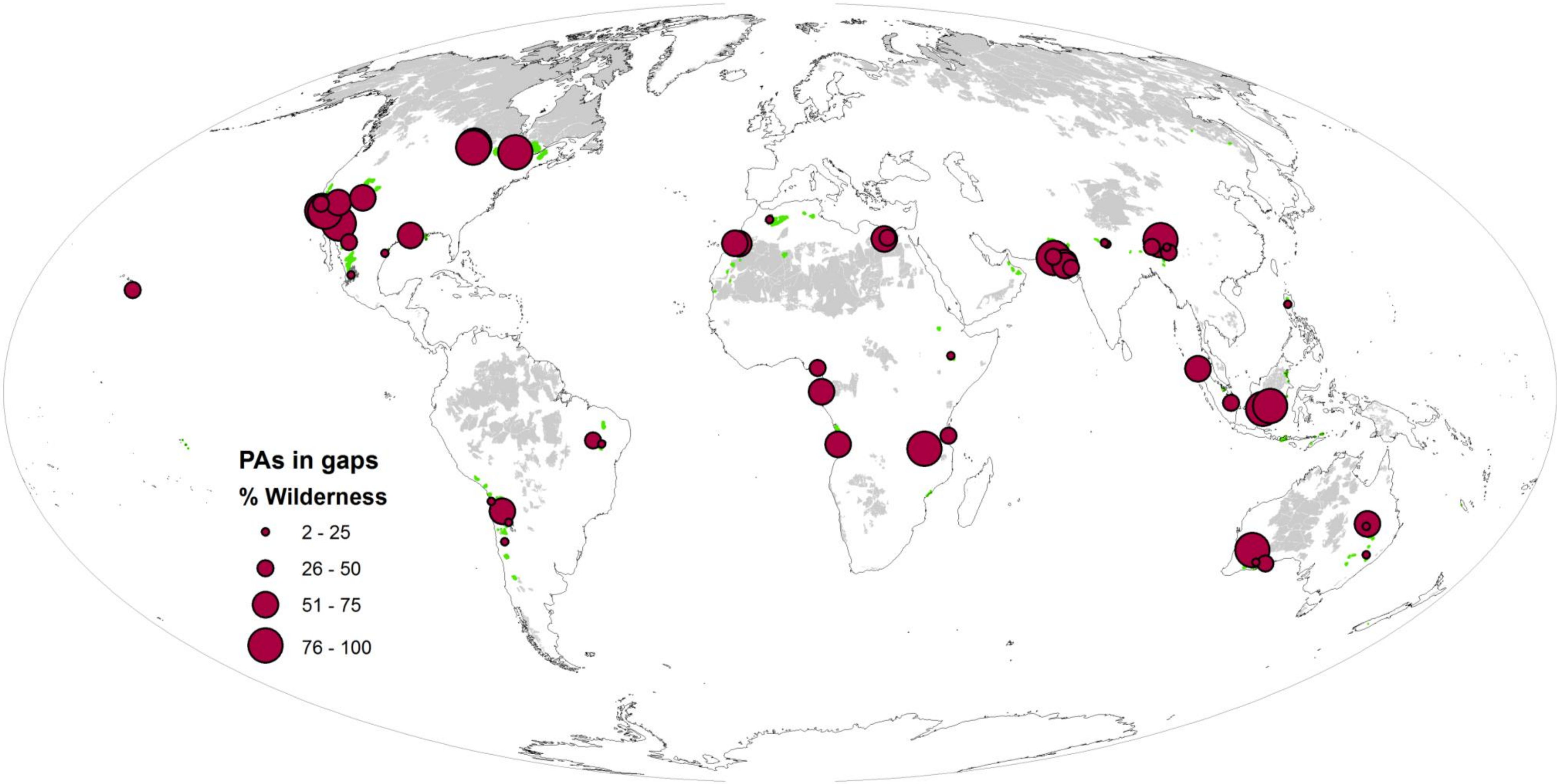
*Notes: * Australasian Tundra shows no wilderness area because the Human Footprint does not extend to the sub-Antarctic islands.*

We identified 1,397 large nationally designated protected areas globally which contain wilderness (Fig 5.2). In 840 of these, wilderness areas make up >50% of the site. These areas are spread across all biomes and 18 of the 22 biorealms identified as gaps in coverage (see supplementary Table S5.2), and can be considered a preliminary set of potential candidate sites for further World Heritage assessment (Fig 5.3). Some examples of protected areas with a high level of wilderness area coverage include the *Eduardo Avaroa Andean Fauna Reserve* in Bolivia (3,968 km^2), the *Hukaung Valley Tiger Reserve* in Myanmar (8,852 km^2) and the *Wild Ass Reserve* in the *Little Rann of Kutch* in India (16,541 km^2 combined).

Discussion

Our analysis is the most up-to-date, systematic and comprehensive assessment of terrestrial wilderness area coverage by the World Heritage Convention. We identified which currently designated WHS contribute to protection of wilderness areas, where gaps in coverage exist, and where large protected areas cover wilderness areas. The World Heritage Committee, States Parties to the World Heritage Convention, other governmental and non-governmental actors, and formal advisory bodies such as the International Union for Conservation of Nature (IUCN) can use our results to assess where wilderness areas are contributing to the OUV of WHS but are not explicitly noted in their statements of OUV. This can then inform management decisions for these WHS, be used to re-assess the configuration and boundaries of current WHS, and guide the identification of possible future WHS where wilderness quality contributes to OUV.

Figure 5.3 All protected areas larger than 500 km^2 with wilderness area coverage, found within biorealms (Olson 2001) where there is no wilderness coverage by the World Heritage Convention. PAs are scaled by the percentage of their area which is wilderness area. PAs with good wilderness coverage within these biorealm gaps could potentially be designated as new Natural or Mixed World Heritage Sites if they meet the strict requirements for World Heritage status.



We found that almost all the wilderness areas without WHS have coverage in large nationally designated protected areas, so already receive some level of protection. These places warrant further evaluating for OUV and therefore potential designation on the World Heritage List. A WHS designation is advantageous because States Parties are then able to access additional resources for conserving these sites. For example, financial support can be accessed through the World Heritage Fund and other funding mechanisms, as well as technical support from UNESCO and its Advisory Bodies, including IUCN (Conradin et al. 2014). WHS are also subject to strict additional monitoring by UNESCO and the Advisory Bodies who co-operate with States Parties which helps ensure WHS are well conserved. The IUCN also carries out assessments of the conservation outlook for each WHS which provides important information and impetus for more effective management and conservation of these sites (Osipova et al. 2014, UNESCO 2015). As a result, allocating these sites World Heritage status is regarded as providing an extra layer of protection. In fact, World Heritage status is the only conservation designation for which several major players of the extractive industry have accepted no-go commitments (ICCM 2014, WWF-UK 2015), further demonstrating its importance and high regard.

Designation on the World Heritage List is an exclusively State Party driven process, where countries have to nominate potential sites for consideration by the World Heritage Committee. These places are then subject to a lengthy evaluation process that takes years. This includes site-level expert assessments to confirm whether they meet the strict requirements for OUV (UNESCO 2015), namely one or more of the World Heritage criteria, conditions of integrity, and requirements of management and protection. States Parties can therefore use the information provided in our study to inform future nominations in cases where wilderness contributes to the OUV of a potential WHS. The Convention's Global Strategy calls on States Parties to develop a balanced, representative and credible list of the world's Natural Heritage (UNESCO 2011). Considering that many wilderness areas are an irreplaceable and dwindling natural entity (Watson et al. 2016c), increasing representation of wilderness areas with OUV within WHS aligns well with the aims of the Convention (Kormos et al. 2016). Furthermore, many processes and species (criteria ix and x) can only be conserved in large, ecologically functional wilderness areas (Ripple et al. 2015), making their protection essential to realising the Conventions global strategy.

Some of the protected areas we identify with wilderness area coverage are already on States Parties tentative lists of potential future nominations, a prerequisite for new nominations. States Parties could consider prioritising these places and strengthening proposals with this additional information on wilderness area coverage. For example, the *Bale Mountains National Park* in Ethiopia is a tentative listed site which protects wilderness in *Afrotropic montane grasslands and savannas* (1326km^2). This biorealm is currently a gap in coverage, so designation of the *Bale Mountains National Park* as a WHS is a potential opportunity to improve representation and protection of wilderness areas within the framework of the Convention. Although not falling into any biorealm gaps, the *Hukaung Valley Tiger Reserve* in Myanmar is another tentative listed site which covers large areas of wilderness. We also recommend a revised nomination of *Pimachiowin Aki* in boreal Canada, which includes several large protected areas with substantial wilderness areas from our analysis (i.e. *Atikaki Provincial Park*, *Woodland Caribou Provincial Park* and *Traditional Land Use Planning Areas of Anishinaabeg First Nations*), should be discussed by the World Heritage Committee in 2018.

Whilst potential new WHS are being evaluated and considered by the World Heritage Committee, States Parties could further strengthen their current management practices and protection in those WHS containing wilderness areas. This is particularly important given that the ecological condition of many WHS is declining worldwide as human pressures expand both within their borders and the surrounding landscapes (Osipova et al. 2014, Wang et al. 2014, Allan et al. 2017c), threatening to isolate them and degrade their wilderness quality and ultimately their OUV. Increasing the application and enforcement of the Convention's Operational Guidelines (UNESCO 2015) which do not permit agricultural expansion, extractive industry, or other similar activities to occur within WHS boundaries is an essential first step to improve their conservation.

Where WHS depend on wilderness quality for their OUV, for example to support wide ranging or migratory species (Chester et al. 2012, Ripple et al. 2014), maintaining connectivity is key (Kormos et al. 2016). This can be achieved efficiently by using existing tools within the World Heritage Convention such as buffering or expanding WHS boundaries (Kormos et al. 2016). If expanding a site's boundaries does not impact its OUV and enhances the site, they are regarded as minor modifications and are subject to an accelerated review process which is much faster than designating a new WHS (UNESCO 2015). Adding or expanding a buffer zone, is treated as a minor modification and can also be accomplished relatively quickly. WHS are expected to have official buffer zones which are managed to support the functioning and protection of the WHS itself (UNESCO 2015).

Where migratory species rely on multiple non-adjacent WHS, the creation of agreements between the sites can ensure management and protection efforts are co-ordinated. This is a tool which has only been applied on one occasion, but can potentially be leveraged to help protect wilderness values (Kormos et al. 2016).

Some WHS which are well known for their wilderness quality such as the *Selous Game Reserve* in Tanzania and *Yellowstone National Park* in USA were not identified in our analysis. This is almost certainly due to how we defined wilderness areas, suggesting that our computer-based analysis is best used in conjunction with site-based assessments and that further research is needed into the significance of different thresholds in wilderness quality. Quantitatively measuring wilderness allows for flexibility in the thresholds used to map wilderness areas (Mackey et al. 1998). As noted above, although these thresholds are arbitrary, the thresholds used here are likely to capture large scale ecological processes, and have been used by others to identify wilderness areas (Mittermeier et al. 2003, Kormos et al. 2016, Watson et al. 2016c), and are consistent with the parameter values used for identifying intact ecological communities in the IUCN Standard for Key Biodiversity Areas (IUCN 2016).

The definition of wilderness quality used here was less strict than Watson et al. (2016c) who defined it as any area completely free of human pressure using global datasets (i.e., Human Footprint score = zero). Our definition was more useful for the regional-scaled analysis that was the focus of this study and that allowed the areas with the relatively highest wilderness quality to be identified within each biorealm. Despite this attempt to enable a more appropriate grain to the analysis, the *Selous Game Reserve* fell outside the 10% threshold for the *Afrotropic tropical and subtropical grasslands and savannas* biorealm, and *Yellowstone National Park* has roads fragmenting its contiguous area so did not meet the 10,000km^2 area threshold and was not one of the ten largest contiguous areas for the *Nearctic temperate coniferous forest* biorealm. Both of these WHS still maintain most of their large mammals, wide-ranging species, and are relatively free of human pressure (UNESCO 2016b, g), making their exclusion an artifact of our study design, and not necessarily a reflection on their wilderness quality.

The World Heritage Convention could better achieve its objectives by formally recognizing the contribution and significance of wilderness quality to the OUV of many WHS. This could for example be done by amending the Convention’s Operational Guidelines to include the word wilderness in the Natural World Heritage criteria and/or conditions of integrity and management and protection. Although it may be a difficult and lengthy process for the World Heritage Committee to agree on this, such evolution of the guidelines in light of evolving conservation thinking has occurred in the past (Bertzky et al. 2013, UNESCO 2015). Since wilderness quality contributes to the OUV of one quarter of currently designated WHS the argument for including wilderness in the criteria for OUV is compelling, and certainly warrants discussion. Official acknowledgement of this kind would in turn raise the profile of wilderness conservation more widely in other multilateral environmental agreements and promote recognition of the importance of wilderness protection in international policies (Watson et al. 2016c). At the very least tentatively listed sites, new World Heritage nominations and current monitoring of WHS could account for the significance of wilderness areas and wilderness quality to OUV.

World Heritage protection of wilderness areas will also generate co-benefits which extend beyond natural heritage conservation. For example, averting the destruction of carbon rich ecosystems such as the last intact forests could prevent the release of substantial CO_2 emissions, and play a key role in the fight against climate change (Lovejoy 2016). Moreover, it is well accepted that proactively conserving intact ecosystems is the most important adaptation for biodiversity and human-kind (Watson et al. 2013, Martin and Watson 2016, Scheffers et al. 2016). World Heritage status can also bring multiple opportunities for sustainable development (Conradin et al. 2014). World Heritage protection of wilderness could therefore contribute to achieving both the environmental and economic objectives such as the United Nations Sustainable Development Goals identified by the United Nations General Assembly (United Nations 2015a). Such a broad base of co-benefits could also serve as a basis for States Parties to start leveraging funding for the conservation of wilderness areas from international donors and programs such as the Global Environment Facility or the Critical Ecosystems Partnership Fund. We therefore conclude that the World Heritage Convention could better achieve its objectives and make a substantial contribution to the conservation of wilderness areas through at least four avenues: By formally acknowledging the contribution of wilderness areas to OUV which would raise the profile of wilderness conservation worldwide, by strengthening current protection of wilderness within WHS, by expanding or re-configuring current WHS, and by designating new WHS which adds an important layer of protection and recognition to large wilderness areas with OUV.

CHAPTER 6 Patterns of forest loss in one of Africa's last remaining wilderness areas: Niassa National Reserve (Northern Mozambique)

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Abstract

Niassa National Reserve (NNR), located in northern Mozambique, spans 42,300km^2 and supports large populations of endangered fauna including savannah elephants (*Loxodonta africana*), lions (*Panthera leo*) and wild dogs (*Lycaon pictus*). The Reserve also sustains the livelihoods of > 40,000 people who rely on the enclosed natural resources to meet many of their subsistence needs. Accurately monitoring fine scale spatial and temporal trends in land-use and tree-cover is increasingly used for monitoring the ecological state of important conservation areas. Here we provide essential information on land-use changes in and around NNR to support ongoing conservation efforts in the region. We examined patterns of forest and woodland loss in NNR between 2001 and 2014 using high resolution maps of global tree-cover change, and compare this with changes in the wider northern Mozambique region. We found that the Reserve lost 108 km^2 of forest (amounting to 0.9% of NNR's 11,970 km^2 aggregated forest and woodland extent), with the majority (89 km^2) of forest loss occurring as a result of expanding agriculture around the two largest village settlements and agricultural practices along some of the main roads into the Reserve. Although this loss was substantial, it is much lower than changes in the surrounding region, with the adjacent districts and four Northern Provinces losing 200 km^2 (3.2%) and 6,594 km^2 (5.7%) of their respective forest extents. We found NNR's diverse Miombo ecosystems are still intact and could support very large mega-faunal assemblages which is a positive story for effective conservation in this region. Given recent calls to proactively 'upgrade' protected areas that have great potential to contribute to biodiversity and broader societal objectives, investment in ensuring the long-term success of NNR is an obvious global conservation priority.

Introduction

Niassa National Reserve (NNR) is Mozambique's largest protected area, spanning 42,300km^2, and is one of Africa's most iconic wilderness areas (Mittermeier et al. 2003). It is situated in far northern Mozambique which is one of the least biologically explored places in Africa (Ryan et al. 2010). NNR is connected to the Selous Game Reserve in Tanzania to its north, via the Selous-Niassa corridor, which permits wildlife to move between the two Reserves (Mpanduji et al. 2002, Mpanduji and Ngomello 2007). Together, the NNR and the Selous Game Reserve form a massive ~150,000 km^2 trans-frontier conservation area (Noe 2015). The region is renowned for having the largest and best preserved tracts of Miombo woodland left in Africa (Mayaux et al. 2004, Ribeiro et al. 2008a, Soto 2009, Maquia et al. 2013), which are globally important for carbon storage and sequestration (Ribeiro et al. 2013, Lupala et al. 2014). These woodlands also provide critical habitat for many of Africa's wide ranging species and threatened mega-fauna (Mpanduji et al. 2002, Riggio et al. 2013, Bauer et al. 2015), supporting Mozambique's largest populations of savannah elephants (*Loxodonta Africana*), lions (*Panthera Leo*), critically endangered wild dogs (*Lycaon pictus*), and a broad assemblage of Miombo species (Begg and Begg 2007, Begg and Begg 2012, Booth and Dunham 2014, Grossmann et al. 2014).

NNR also supports a growing population of approximately 40,000 people who live within the Reserve boundaries in two towns Mecula and Mavago and ~40 smaller scattered villages. These people experience very high levels of poverty and their access to infrastructure and social services is limited (Cunliffe et al. 2009, Jorge et al. 2013). They therefore depend heavily on NNR's biodiversity and resources for their livelihood and subsistence needs (Campbell et al. 1996, Cunliffe et al. 2009). The principle livelihood activity has been shifting slash-and-burn agriculture (Cunliffe et al. 2009), which is legal under certain conditions in National Reserves under Mozambican law. However, this agriculture is both expanding and becoming more static as settlements become more established, and the resulting land conversion is in opposition to NNR's conservation objectives (SDGRN 2006, Cunliffe et al. 2009). Other examples of legal livelihood activities in NNR include fishing and honey gathering, whilst many households also rely on illegal subsistence bush-meat hunting, and some earn cash from artisanal mining and other illegal activities (e.g. ivory poaching, logging). The Reserve management authority allocates a yearly wildlife quota for communities to hunt, and also share 16% of the total revenue generated through commercial photographic and hunting tourism directly with communities through Community-based Natural Resource Management Committees (Jorge et al. 2013). This community engagement is based on growing evidence that well managed protected areas can reduce poverty, improve rural livelihoods and promote peace and stability (Ferraro et al. 2011, Naughton-Treves et al. 2011, Maekawa et al. 2013).

Since the end of the Mozambican civil war in 1992, there has been a dramatic increase in land conversion for agriculture across northern Mozambique, as people returned to rural lands which they had previously abandoned (Temundo 2004, Temudo and Silva 2012). This is a well-established post conflict pattern and the consequences for biodiversity can be devastating (McNeely 2003, Negret et al. 2017). Mozambique's human population is also growing rapidly at a rate of ~3% per year, putting increasing pressure on the country's natural resources (Temudo and Silva 2012, Crist et al. 2017). Likewise, the human population within NNR has grown at a similar rate (INE 2008b, a), compounded by immigration from outsiders attracted by NNR's biodiversity, other resources and space for agricultural expansion (Grossmann et al. 2014, Niassa Carnivore Project 2015). There are concerns that populations of many wildlife species in NNR, which had been steadily increasing since the end of the civil war, are being impacted by increasing human pressure (Grossmann et al. 2014). Anthropogenic conversion of intact vegetation, or habitat loss, is one of major drivers of species extinctions globally (Fischer and Lindenmayer 2007, Maxwell et al. 2016), followed closely by overhunting (Tranquilli et al. 2014, Maxwell et al. 2016), both of which pose an immediate threat to NNR's biodiversity and are a major challenge for NNR's management.

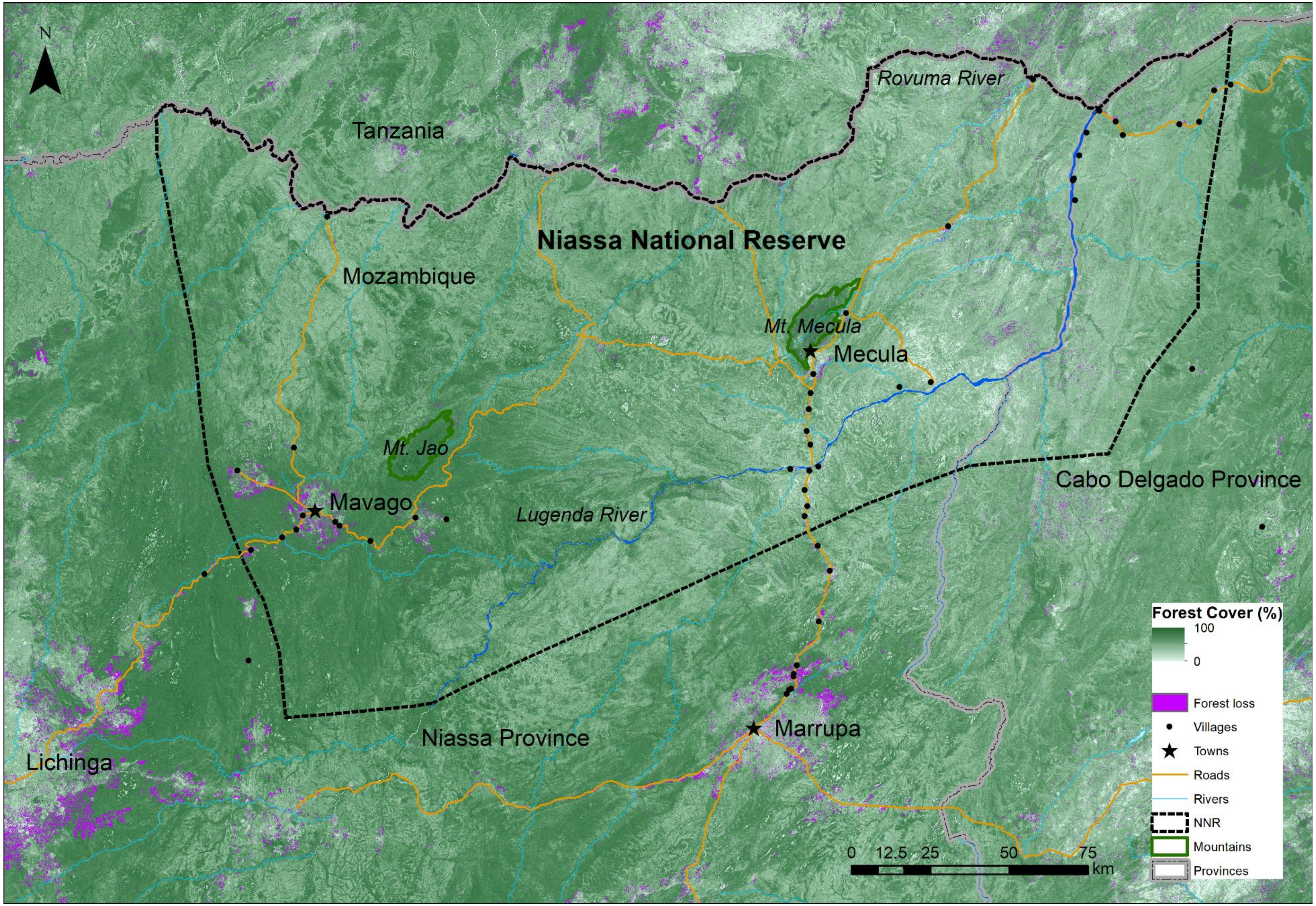
Accurately monitoring fine scale spatial and temporal trends in land-use and tree-cover is increasingly used for monitoring the ecological state of important conservation areas (Nagendra et al. 2013, Tracewski et al. 2016, Allan et al. 2017c). This provides crucial information for conservation planning since it identifies where biodiversity is likely to be threatened and where management actions should be targeted (Turner et al. 2003, Tracewski et al. 2016). However, northern Mozambique is particularly data-poor. Previous efforts to map land-use changes and tree cover in and around the NNR are out-dated (Desmet 2004, Games 2004), temporally static (Ganzin et al. 2010, Prin et al. 2014), or have focussed on carbon and fire dynamics (Ribeiro et al. 2008b, Ribeiro et al. 2013). There is a clear need for more up-to-date information to support conservation decision making.

We aim to address this gap by analysing patterns of forest and woodland loss (hereafter forest loss) in NNR between the years 2001 and 2014 using high resolution maps of global tree-cover (Hansen et al. 2013b). We identify which areas in NNR have suffered the greatest forest loss, and which areas are faring well with limited negative changes to this key component of their ecological integrity. We also compare our findings for NNR with patterns of forest loss across all of northern Mozambique to provide regional context. Key findings from this study can be used by the Reserve management to inform conservation decision making. We also hope to draw more research attention to an understudied region that is globally important for biodiversity conservation.

Study Site

NNR is a socio-economically sensitive and politically complex region; it is in northernmost Mozambique bordering Tanzania, and extends across two provinces; Cabo Delgado and Niassa, and nine administrative districts (Figure 6.1). NNR was officially proclaimed in 1954, but then abandoned between 1975 and 1992 during Mozambique's civil conflict. Once a peace accord was signed, the Mozambican government made a series of agreements with private companies and non-governmental organizations (NGOs) to manage NNR (SDGRN 2006). Since October 2012, The Wildlife Conservation Society has been co-managing NNR with the National Administration for Conservation Areas in Mozambique to secure the long-term future of NNR. The Reserve is divided into 18 management blocks of which 16 can be leased as concessions by private concessionaires. Sustainable use of wildlife is permitted within NNR, and eight concessions are currently leased for hunting tourism and two are vacant. One concession is informally designated for community use, four are leased for photo tourism and one is vacant. Two blocks are strictly protected for biodiversity conservation.

Figure 6.1 The extent of forest loss in Niassa National Reserve



NNR has a tropical sub-humid climate, with mean monthly temperatures between 20 and 30 degrees Celsius. The wet season runs from November to April and the mean annual rainfall is 900mm. Rainfall increases from east to west (800mm – 1,200mm) across NNR, as does the altitude (200m – 1,400m above sea level). The highlands in the west are well forested and continue beyond NNR's boundaries forming the watershed for its two major rivers; the Rovuma and the Lugenda. Both rivers have strong perennial flows that are key for supporting NNR's biodiversity and people. There are two major peaks in the Reserve, Mount Jao (1,200m) and Mt Mecula (2,000m), which contain important protected montane forests in Mozambique and are centres of high diversity in the Miombo belt. The habitat in the rest of NNR (72%) is predominantly Miombo woodland dominated by *Brachystegia* and *Julbernardia* tree species (Mayaux et al. 2004, Ribeiro et al. 2008a). Vegetation dynamics are largely driven by the rainfall gradient across NNR, and a complex interaction between fire (mainly anthropogenic) and elephants, whose destructive herbivory can increase fuel loads and fire intensity (Ribeiro et al. 2008b, Ribeiro et al. 2013).

Methods

We examined patterns of forest loss and gain in NNR and northern Mozambique between 2001 and 2014 using high spatial resolution maps of global tree-cover (Hansen et al. 2013b). The Global Forest Change dataset is the most accurate representation of temporal forest loss available (McRoberts et al. 2016). We defined forest cover as vegetation taller than 5m, and forest loss as the complete removal of canopy cover at a 30m resolution. Data was extracted and processed in the Google Earth Engine (http://earthengine.google.org/), a cloud platform for earth-observation data analysis. We summed the extent of year by year forest loss between 2001 and 2014 to calculate the total extent of forest loss in NNR during this time period, and present this as a percentage of the total forest extent in 2000. We also analysed the total gain in forest cover extent between the years 2001 and 2012. The forest cover gain data is not available in year by year time series, and cannot be compared directly with the forest loss data since they were developed using different methodologies (Hansen et al. 2013b). We adapted JavaScript code developed by (Tracewski et al. 2016) for analyzing forest cover data within specified spatial zones, which is freely available online (https://github.com/RSPB/IBA). Forest loss indices were aggregated to the district and provincial scales as they provide useful units representing political organisational entities and hence management levels. To provide context we compare trends in forest cover in NNR to trends in the surrounding landscapes, which we defined as 1) the 26 districts directly adjacent to NNR, and more broadly as 2) the four northern provinces of Mozambique

(Niassa, Cabo Delgado, Nampula and Zambezia). We did not control for landscape or ecological characteristics in our analyses.

Results

We found that the total area of forest lost inside NNR between 2001 and 2014 was 108 km^2, amounting to 0.9% of the 11,971 km^2 of NNR's aggregated forest extent in the year 2000. The majority of forest was lost around the towns of Mecula and Mavago where 41.4 km^2 (0.9%) and 47.5 km^2 (4%) of forest cover was cleared respectively, primarily for agricultural purposes (Figures 6.2 and 6.3). Forest cover was also lost along the main Marrupa-Mecula road leading into the centre of NNR, where communities practice shifting agriculture, and in the north-eastern corner of the Reserve near Negomano. The direction of the shifting agriculture was predominantly from NNR's boundaries toward its center along main roads (Figure 6.1). The overall annual average of forest loss in NNR remained fairly consistent across the 12 years studied, with peaks occurring in 2008 - 2009 and 2013 (Figure 6.4).

Forest loss in NNR was much lower than in the surrounding landscape. The 26 districts directly adjacent to NNR (in the provinces of Niassa and Cabo Delgado) lost an average of 4.4% of their forest cover between 2001 and 2014. The districts of Lichinga, Mueda and Nangade suffered the most, losing 187 km^2 (9.5%), 170 km^2 (4%) and 134 km^2 (10.1%) of their forest cover respectively during the study period (Table 6.1). Likewise, the northern Mozambican provinces of Niassa, Cabo Delgado, Nampula and Zambezia (excluding NNR) lost a total of 6,594 km^2 of forest cover amounting to 5.7% of the 116,010 km^2 of forest cover in the region in the year 2000. The overall rate of forest loss in the provinces and districts of northern Mozambique increased over the study period, with peaks in 2008 and 2013 (Figure 6.3).

We found that the total area of forest gain within NNR between 2001 and 2014 was negligible, amounting to 1.1 km^2, which equates to 0.01% of NNR's total aggregated forest extent and 1% of the forest lost during the time period. Forest gain in NNR was also low compared to gain in the districts surrounding NNR which amounted to a more substantial 154 km^2 (0.3% of forest extent, 9% of the forest extent lost), and in the northern provinces of Mozambique which amounted to 573 km^2 (0.5% of forest extent, 8.7% of the forest extent lost).

Figure 6.2 The extent of forest loss around Mecula town and on the Mecula-Marrupa road

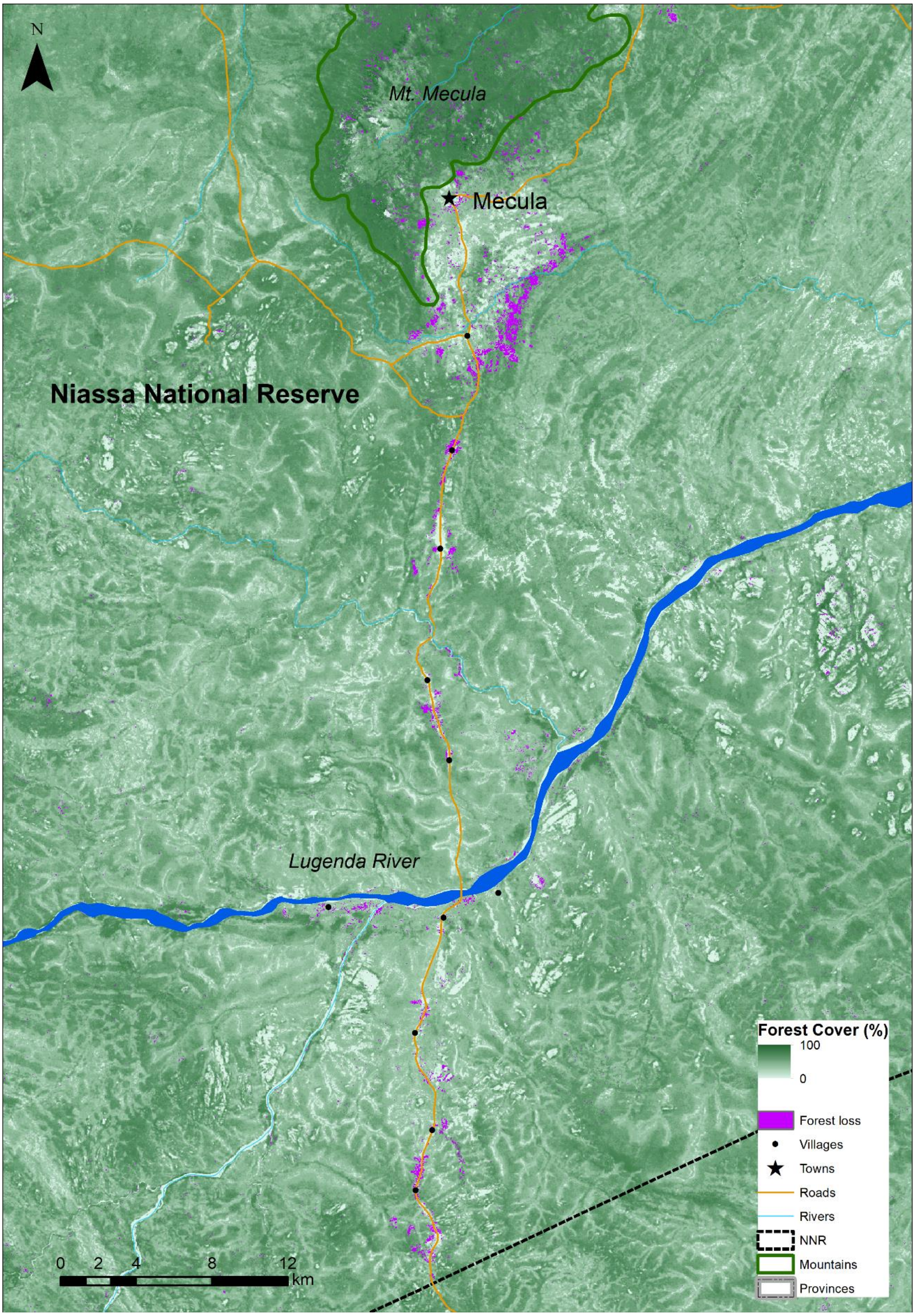

N
Mt. Mecula
Mecula
Niassa National Reserve
Lugenda River
Forest Cover (%)
100
0
Forest loss
Villages
Towns
Roads
Rivers
NNR
Mountains
Provinces
0 2 4 8 12
km

Figure 6.3 The extent of forest loss around Mavago Town.

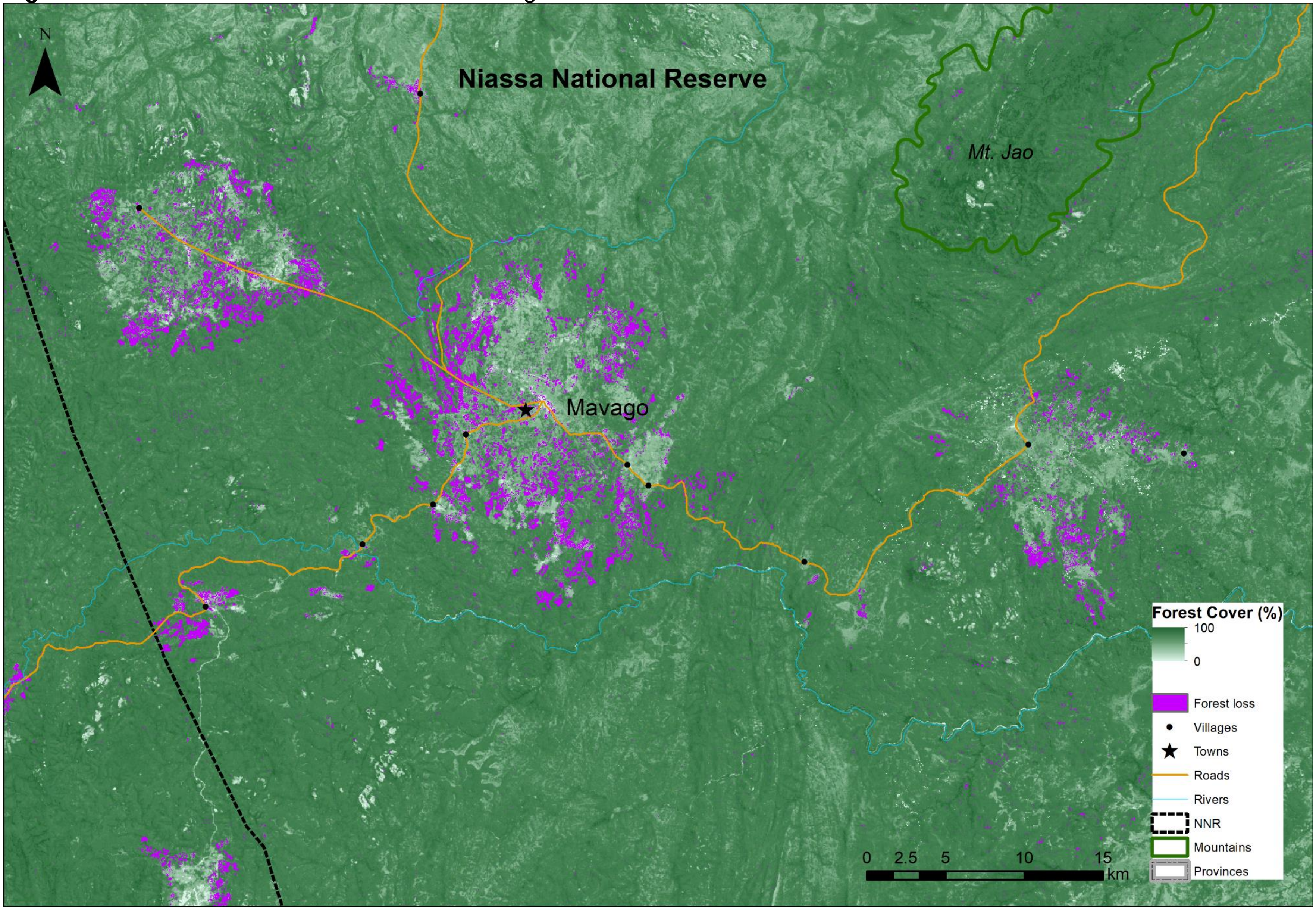
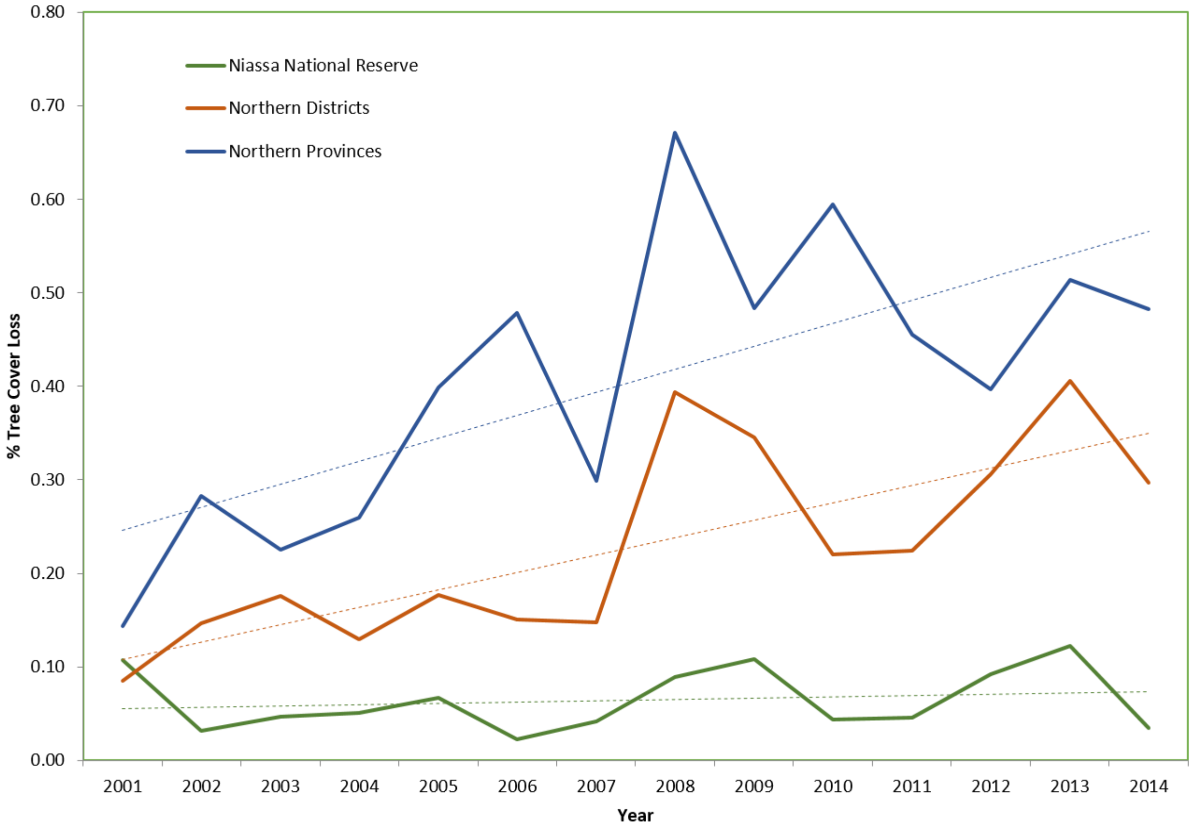


Figure 6.4 The yearly percentage forest loss between 2001 and 2015 for Niassa National Reserve, and Northern Mozambican provinces and districts.



Discussion

Our analysis provides an up-to-date assessment of changes in forest cover in NNR and northern Mozambique between 2001 and 2014 and important baseline information for future conservation planning efforts. We found that NNR lost > 100 km^2 of forest cover amounting to ~ 1% of its aggregated forest extent. This may appear substantial, but is much lower than the 3% of forest cover lost in protected areas globally during the same time period (Heino et al. 2015, Morales-Hidalgo et al. 2015). Our findings are also particularly encouraging in the African context, given deforestation rates on the continent are five times higher than the global average (Tranquilli et al. 2014), and there are many examples of protected areas in Africa losing much more forest cover within their boundaries (Sassen et al. 2013, Bowker et al. 2017).

We found that forest loss was higher in the landscapes surrounding NNR, with some adjacent districts losing up to 10% of their forest extent. This suggests that NNR is performing relatively well at limiting forest loss within its boundaries given external pressure (Bruner 2001), and supports assessments suggesting protected areas are-

Table 6.1 The total aggregated forest extent (km2), total amount of forest loss and gain (km2) between 2001 and 2014, and the percentage of forest loss and gain in Niassa National Reserve, and the districts and provinces of Northern Mozambique.

	Tree extent (km2)	**Tree-loss (km2)**	**% Tree loss**	**Tree-gain (km2)**	**Tree gain as % of loss**
Niassa National Reserve	11970.9	108.1	0.9	1.1	1.0
Districts					
Ancuabe	1823.6	96.3	5.3	6.3	6.6
Balama	1466.7	18.0	1.2	0.2	1.2
Chiure	1190.5	55.4	4.7	1.6	3.0
Lago	2456.0	48.6	2.0	0.9	1.9
Lichinga	1979.5	187.0	9.4	0.9	0.5
Macomia	1419.5	111.0	7.8	26.7	24.1
Majune	4348.5	39.2	0.9	0.6	1.6
Mandimba	950.1	38.7	4.1	0.1	0.3
Marrupa	5175.3	82.0	1.6	0.8	1.0
Maua	2452.0	39.1	1.6	0.4	0.9
Mavago*	3300.7	47.5	1.4	0.4	0.8
Mecanhelas	323.1	22.5	7.0	0.0	0.2
Mecula*	4471.3	41.4	0.9	0.5	1.1
Meluco	2381.8	66.8	2.8	10.4	15.6
Metarica	1506.6	22.7	1.5	0.3	1.3
Mocimboa da Praia	517.8	49.0	9.5	9.4	19.3
Montepuez	4729.5	64.1	1.4	0.8	1.3

Mueda	4075.0	169.9	4.2	39.1	23.0
Muembe	2640.5	66.4	2.5	0.5	0.7
Muidumbe	1051.0	70.6	6.7	19.2	27.3
Namuno	1556.7	66.8	4.3	1.2	1.8
Nangade	1322.9	133.6	10.1	22.3	16.7
N'gauma	1027.1	79.9	7.8	0.2	0.3
Nipepe	1399.4	20.6	1.5	0.2	0.9
Palma	341.9	17.6	5.2	3.1	17.4
Quisanga	331.2	16.7	5.0	7.6	45.7
Sanga	4270.7	43.0	1.0	0.7	1.7
Provinces					
Nampula	21231.7	1705.8	8.0	148.8	8.7
Zambezi	40741.3	2758.4	6.8	175	6.3
Niassa**	28060.7	759	2.7	5.9	0.8
Cabo Delgado**	25977.1	1371.5	5.3	243	17.7

***** Districts entirely in the Reserve **Provinces overlapping the Reserve

-effectively conserving habitat and biodiversity (Geldmann et al. 2013, Barnes et al. 2016). However, because we did not control for landscape characteristics ("matching") there is a possibility we are overestimating the effect of protection (Joppa and Pfaff 2010, Joppa and Pfaff 2011, Geldmann et al. 2013). Although our results do confirm a well-known pattern that protected areas on the African continent and globally are becoming increasingly isolated by land clearing beyond their boundaries (DeFries et al. 2005, Newmark 2008, Bailey et al. 2016). This is concerning since degradation around a protected area strongly predisposes it to future degradation within its borders (Laurance et al. 2012).

We found that the majority of forest loss in NNR occurred around the two largest towns of Mecula and Mavago, where the majority of NNR's human population resides. Since forest loss is locally restricted, NNR's management can target actions to these high risk areas and engage with the local communities. The communities have the right to continue residing within NNR in accordance with Mozambican law, and NNR's management team and concession holders are already working closely with many of them to build connections and interdependencies. For example, the Reserve Management Authority employs between 75-80% of its 150 staff from local villages in NNR, and Mariri concession block employs 80% of its team from local villages in NNR.

Local communities were also engaged during the development of an updated management plan for NNR, and helped define NNR's vision for the next decade. NNR's new management plan, which will run from 2017 to 2027, is the first to explicitly acknowledge local community members as key stakeholders and partners in NNR's future, and to zone sections of the Reserve for community use and development. There are also micro-zoning initiatives planned and underway in imminently threatened areas to try and control agricultural sprawl as human pressure increases. These actions alone will not protect NNR's biodiversity, but is a major step towards repairing the historically fractured relationship between NNR's communities and management. Through stronger collaborations with public-health professionals and social scientists NNR's management can also help ensure human-development goals and the communities' socio-economic aspirations are met. This could help decrease the anthropogenic pressure being placed on NNR's wildlife and habitats, which in turn should translate into increased revenue to communities, since higher commercial and community hunting quotas will be possible.

Our analysis has several caveats worthy of discussion. There are limitations to satellite derived estimates of tree-cover, such as lower accuracy in more arid places, and an inability to distinguish between ecologically valuable forest compositions and commercially valuable forest stands, all of which have been well discussed (Hansen et al. 2013a, Achard et al. 2014, Tropek et al. 2014). Despite its limitations, the Global Forest Change dataset is still considered the most accurate global representation of temporal forest loss available (McRoberts et al. 2016, Gross et al. 2017). We recommend particular caution when interpreting the forest gain data since we cannot be certain that it is the natural regrowth of ecologically valuable vegetation. For example, in Lichinga district there are forestry plantations, and in the districts along the coast there are coconut plantations which could be responsible for some of the gains we recorded outside of NNR. We were surprised to find negligible gain in forest extent within NNR but suspect that fields are not being vacated for the 20 – 30 years required for Miombo woodland to mature (Jew et al. 2016). No commercial forestry is known to occur within NNR.

A second caveat is that we cannot infer the exact causes of forest loss from the data. The patterns of forest loss we identified within NNR – along roads and around villages – strongly suggest that this is the result of anthropogenic clearing to meet local subsistence needs, which has been confirmed by NNR's management who have already surveyed many of the areas where large extents of forest cover were lost. An important extension of this work would be to model and explore the drivers of forest loss in NNR and northern Mozambique. There are also many anthropogenic threats and disturbances beyond habitat loss which are affecting NNR's ecological state and biodiversity, such as commercial poaching, overhunting, wildfires, climate change, artisanal mining and selective logging, which our analysis does not capture (Barlow et al. 2016, Maxwell et al. 2016) (Barlow et al., 2016; Maxwell et al., 2016). Bush-meat hunting using snares, which kill indiscriminately, is occurring in NNR, as is the poisoning of large carnivores such as lions and leopards for their body parts (Niassa Carnivore Project 2015). The lion population in NNR is estimated at 800 individuals but is decreasing in localized areas, with some places now completely devoid of carnivores, which could have serious cascading ecological effects (Ripple et al. 2014). Similarly, between 2011 and 2014 NNR's elephant population declined by 63% to an estimated 4,440 individuals in 2014, driven by illegal ivory poaching (Booth and Dunham 2014, Grossmann et al. 2014). This equates to an estimated loss of just over 7,500 animals – one of the most catastrophic declines on the continent (Chase et al. 2016). Other emerging threats to NNR's wildlife, which could also have negative impacts on forest cover, include artisanal mining and charcoal production (Papworth et al. 2017). The Global Forest Change dataset is updated regularly so continued monitoring can ensure emerging threats are identified and managed as early as possible but there also needs to be additional monitoring efforts and action to secure NNR's species in the long-term.

NNR's wildlife is clearly in danger but it is encouraging that the majority of NNR's Miombo woodland habitat is intact and could support large populations of mega-fauna. There have been recent calls for increased investment in upgrading protected areas that have high but currently un-realized potential both for conservation and communities (Pringle 2017). By upgrading protected areas, we mean increasing their management effectiveness, while harmonizing them with the needs and aspirations of their constituencies (Pringle 2017). Through increased community engagement, and stronger management action against key threats to biodiversity, we suggest NNR could become a flagship for such efforts. NNR has already been identified as a critical protected area for continent-wide lion recovery efforts because it could support well over 1,000 individuals (Lindsey et al. 2017). NNR also has the potential to support approximately 50,000 elephants which is more than ten times its current population (Robson et al. 2017). Residual wildlife populations are still large enough that they could recover naturally assuming levels of persecution decrease and threats are managed. Given the potentially substantial benefits to biodiversity conservation and broader societal goals, investing in the effective management of NNR is an obvious global conservation priority.

CHAPTER 7 Conclusion

Overview

The overarching aim of this thesis was to address key questions relevant to conserving wilderness areas and their immense values (**chapter 1**), with a particular focus on biodiversity conservation. I created the first temporally inter-comparable global maps of terrestrial wilderness areas (**chapter 2**), which enabled an analysis of wilderness loss through time (**appendix 2**). I then analysed where the human pressures responsible for wilderness loss are impacting threatened species globally (**chapter 3**) and Natural World Heritage Sites, which are important places set aside to protect threatened species and wilderness (**chapter 4**). I then analysed opportunities for the World Heritage Convention to make a greater contribution to wilderness conservation (**chapter 5**), and present a case study of a protected area that potentially warrants World Heritage status based on its wilderness values (**chapter 6**). In this final chapter, I provide an overview of the main findings and discuss their significance for wilderness and biodiversity conservation. I also present some of the conclusions that emerge from looking at this thesis as a whole and discuss some limitations of the research. Finally, I outline some future research opportunities that could lead to improvements in the field.

Scientific advancements and conservation applications

When I commenced this thesis, information on the location, threat, and protection status of wilderness areas were limited. Recognising that this was a major barrier to wilderness conservation, in **chapter 2,** I used the updated Human Footprint dataset (**appendix 1**) to systematically identify global terrestrial wilderness areas. I created two sets of maps for the years 1993 and 2009, including maps of ‘pressure free lands’, and regionally representative wilderness maps following the ‘Last of the Wild’ methodology (Sanderson et al. 2002). These datasets are the most up to date products available, and are the first temporally-inter comparable global maps of wilderness areas developed to date.

The method I used to map wilderness (identifying pressure free areas) is similar to recent efforts identifying global roadless areas (Ibisch et al. 2016), and the location of marine wilderness areas (Jones et al. 2018a). The work also complements efforts to identify intact forest landscapes, which followed a different approach that utilised satellite imagery data to assess direct human induced forest structural alterations and fragmentation (Potapov et al. 2017). Determining the most expedient or accurate way to map wilderness is still an unsolved problem. In this thesis I have demonstrated one approach for mapping wilderness, that can provide a stepping stone for future efforts. The maps presented here have also proved useful for multiple analyses that can help inform global conservation.

The wilderness maps enabled a collaborative effort led by Watson et al. (2016c) (**appendix 2**), where we analysed changes in wilderness extent and found that ten percent (3.3 million square kilometres) of Earth's wilderness was lost between 1993 and 2009. To put this in context, it is an area greater in size than India, and the rate of loss is roughly four times the rate of global forest loss during the same time period (Keenan et al. 2015). A recent analysis of the location of marine wilderness areas found that they have almost completely retracted to the polar regions, with only 13% of the ocean still pressure free (Jones et al. 2018a). This illustrates that wilderness areas on land, and in the sea, are far more imperilled than was previously thought. Time is clearly running out to secure the future of Earth's wild places (Watson et al. 2018b). Concerningly, many of the most widespread wilderness losses occurred in the most biodiverse regions on Earth including the Amazon and Central African rainforests, which lost >30% and 14% of their wilderness respectively. Efforts to conserve wilderness areas within protected areas have also lagged behind the rate of loss, which was double the rate of protection between 1993 and 2009.

In **chapter 3,** I analysed where the human pressures responsible for wilderness loss are having the greatest impacts on threatened biodiversity. I did this by developing a novel method to connect spatial data on the eight threatening processes in the human footprint (**appendix 1**) with the geographic distributions of over 5,000 threatened terrestrial vertebrates. I then analysed the extent of species-specific threats within those species' ranges. A major innovation of this work is that the methodological framework allows us to extend beyond just analysing human pressures, which do not account for species type and their sensitivity to threats, to analysing realised impacts on individual species. By filtering out the human activities that co-occur with species but do not cause them harm, from the activities directly responsible for species declines and increased risk of extinction, I identify realised impacts on biodiversity. This framework offers a tool for defining strategies to directly mitigate the threats endangering a species. There are also opportunities to extend the methodological approach presented here to the marine and freshwater realms, where high resolution threat maps have been developed following somewhat similar methods to the terrestrial human footprint (Vorosmarty et al. 2010, Halpern et al. 2015, Venter et al. 2016c).

The results of **chapter 3** show that humans are impacting threatened species across 84% of Earth's surface, and identify hotspots of impacted and unimpacted species richness (refugia). One of the most striking findings is that humans are impacting a proportion of the species assemblage everywhere that both a human pressure and a threatened species co-occur. This suggests that there are currently no examples of perfect co-existence between humans and the entire assemblage of species in a given area – that is, anywhere that human pressures exist, those pressures negatively impact a threatened species. This further highlights the importance of protecting wilderness areas (pressure free lands), which serve as key refuges where biodiversity will likely persist. The results in **chapter 3** are consistent with other recent studies that modelled the impacts of human land-use on species assemblages identifying widespread declines in species richness and abundance (Newbold et al. 2014, Newbold et al. 2016).

It is important to note that the maps of human impacts on threatened vertebrates (**chapter 3**) are not directly comparable with the maps of wilderness areas or human pressure (**chapter 2, appendix 1**) because they were developed at different spatial resolutions (human impacts 30 km^2; wilderness and human pressure 1km^2). This difference in resolution explains why human impacts extend across 84% of Earth's surface, whilst human pressures only extend across 75% (Venter et al. 2016c). Although the human footprint data is available at 1km^2 resolution globally, 30 km^2 is a more appropriate resolution when working with IUCN species range maps for reducing the effects of commission errors (where species are thought to be present but are not) (Di Marco et al. 2016c). The human impact maps should be interpreted in the context of this spatial grain; however, they still represent the current best estimate of human impacts on threatened terrestrial vertebrates globally, and the methodological framework is a conceptual advance for the field of threat mapping. I intend to refine the resolution of this work in the future as data precision improves. In particular, the global mammal assessment team at Sapienza University in Rome are currently creating revised distribution models for all mammals that when ready could be directly used in an update and refining of **Chapter 3**.

Both the wilderness maps and the underpinning Human Footprint data were rigorously validated for accuracy by using high-resolution (0.5m) satellite images to visually confirm if human pressures were present or absent across 3114 randomly selected points. Although both datasets exhibit an excellent degree of accuracy, they have several limitations arising from the data they do not include (Hulme 2018, Jones et al. 2018c). As briefly mentioned in several of the chapter discussions, the human footprint is not inclusive of all possible threats to biodiversity and is likely an underestimate of human pressure. This means the human impact maps are likely underestimates of true impact, and conversely, that the wilderness maps are likely overestimates of wilderness extent. Indeed, a quick glance at google earth imagery shows that many places that were still wilderness in 2009 have been eroded since then.

The original human footprint data (Sanderson et al. 2002) were previously compared to regional human pressure maps, and the accuracy and resolution of the global datasets for regional analyses is not always appropriate (Leu et al. 2008, Woolmer et al. 2008). For example, in Canada, the global human footprint shows a 16% mean countrywide decline from 1993-2009, due primarily to drops in population pressure as part of the urbanisation process (Venter et al. 2016c). However, this quantification of change does not include pressures that lack globally consistent data such as oil, gas, mining and forestry, all of which have extensive and rapidly changing footprints both in Canada, and many other countries worldwide (Venter et al. 2006, Butt et al. 2013, Harfoot et al. 2018). This highlights the need for a more comprehensive updated human footprint, and downscaled versions of the human footprint that include regionally specific threats to support local or regional analyses and decision making (González-Abraham et al. 2015, Tapia-Armijos et al. 2017).

Other important threats to biodiversity not directly considered in the analyses in this thesis are anthropogenic climate change and invasive species. This is an important limitation to consider because climate change is already significantly impacting species, biomes, and people worldwide (Scheffers et al. 2016). A recent analysis also showed that climate change is already impacting the entire extent of marine wilderness areas (Jones et al. 2018a). To my knowledge, there are no studies assessing the exposure and vulnerability of different terrestrial wilderness areas to climate change, representing an important avenue of future work. Invasive species are a leading cause of extinctions globally (Clavero and García-Berthou 2005) and are a key threat for almost one quarter of endangered species (Maxwell et al. 2016). However, roads and human population density, which are included in the human footprint data, have been documented as good proxies for the presence of invasive species (Hulme 2009, Meunier and Lavoie 2012). Data on invasive species is becoming increasingly available at the regional level. For example, in Australia, there are ongoing efforts to map the extent and density of feral cats and foxes, which are both harmful invasive species. This data could be included in regionally downscaled threat maps shortly (Legge et al. 2017).

However, developing globally standardised data on threatened species will be a major challenge, and is unlikely to be available in the near future.

It is clearly important to capture a more comprehensive range of threats, including climate change and invasive species in future updates of the human footprint data. However, as mentioned in the discussion of **chapter 3**, one of the fundamental ways to manage threats such as climate change or invasive species, is to ensure adequate management of the more easily abatable threats that are captured in the human footprint data (Ripple et al. 2016). By doing this, it is possible to avoid or minimise antagonistic and synergistic interactions between multiple threats (Brook et al. 2008, Mantyka-Pringle et al. 2015, Côté et al. 2016). Therefore, despite not including climate change or invasive species, I am confident the data sets developed in, and underpinning this thesis still provide useful information for conservation practice and policy.

Many of the caveats around the use of global scale remotely sensed datasets such as the human footprint and global forest watch data are noted in the discussion sections of each chapter. Here, I briefly summarise some of the main ones, and discuss some further caveats emerging from using big remotely sensed data sets. Some caveats of the human footprint are that it does not consider all threats to biodiversity (discussed in chapters 3,4 and 7), and we often assume species and taxa will respond equally to different human pressures (discussed in chapter 3). Another challenge is that it is difficult to infer the cause of changes in human pressure or forest loss without developing complex models of the potential predictors, which would involve stand-alone studies of their own (discussed in chapters 4 and 6). Some caveats of the global forest change data include lower accuracy in more arid environments and the inability to distinguish between ecologically valuable forest and plantation forests such as oil palm (discussed in chapters 4 and 6).

The data sets used in this thesis are global in scope, allowing for the broad scale comparisons presented. However, many regional or national datasets likely exist that may be of higher resolution, or more appropriate in a particular local context. An important direction for future research would be to tighten the link between patterns shown in broad scale datasets, and biodiversity responses on the ground. For example, by combining aerial count data on wildlife populations in NNR with forest loss data it would be possible to asses the impact small holder agriculture is having on biodiversity. Similarly, it would be interesting to measure aspects of biodiversity such as species richness, abundance, and intactness of the species assemblage along a gradient of human pressures (beyond the <0 used in the wilderness map) and a range of size thresholds (beyond the 10,000km^2 threshold used) to see what aspects of biodiversity are lost as a landscape transitions away from wilderness.

In **chapter 4**, I carried out the first global assessment of changes in human pressure (data in **appendix 1**) and ecological state across the entire network of WHS. These are the jewels in the crown of the conservation movement, protecting the most outstanding and unique natural areas globally. They protect many wilderness areas and important sites for biodiversity conservation. I found that human pressure had increased in 63% of WHS, and forest loss in 91%, potentially jeopardizing their outstanding natural values. The most concerning finding was that the condition of ~20 sites was deteriorating rapidly, potentially beyond repair. **Chapter 4** extends previous quantitative analyses of the threats to WHS through the use of globally standardized threat data, which allows robust comparisons to be made between regions. The results showed that damage to WHS is occurring across all regions and continents including in many developed nations. I also included a much larger number of WHS than previous studies, analyzing every WHS that had been inscribed prior to 1993 globally.

The findings in **Chapter 4** highlighted several major things; firstly, WHS are much more threatened than was previously thought and many require immediate intervention. Secondly, there is a systemic global failure to prevent harmful human activities occurring within protected areas. This is further supported by a recent collaborative effort with Jones et al. (2018b), where we extend the methodological approach in **chapter 4** to >40,000 protected areas globally finding that one third of protected land is under intense human pressure. One potential reason for this is that conservation targets (such as protected area expansion) are not directly linked to conservation outcomes (Barnes 2015, Barnes et al. 2018). I elaborate on this in the next section where I discuss some of the policy implications of this thesis.

In light of my finding that humans are negatively impacting many WHS, it became important to explore how the WHC could better protect the ecological integrity of WHS and make a greater contribution to wilderness conservation. In **chapter 5**, I used the 'Last of the Wild' maps developed in **chapter 2** to identify which currently designated WHS contribute to the protection of wilderness areas. I also assessed where gaps in the WHC's coverage of wilderness exist, by identifying biorealms (the biogeographic units of the analysis; see **chapter 5** methods) where no wilderness is currently protected in WHS. Within these gaps

I identified >800 large protected areas with excellent (>50% of their area) wilderness coverage. Assuming these protected areas meet the other strict requirements of the World Heritage Convention, they could be designated as WHS for their wilderness values. This represents an opportunity for the World Heritage Convention to take a systematic approach to protecting representative samples of wilderness areas within WHS. I also explored how existing tools within the WHC could be leveraged to improve wilderness conservation. I concluded that the WHC could make a substantial contribution to the conservation of wilderness areas through several avenues, including expanding or re-configuring current WHS boundaries to protect more wilderness. By designating new WHS to add an important layer of protection and recognition to large wilderness areas, and by taking immediate action to strengthen the current protection of wilderness within WHS. This is crucial given the finding of **chapter 4**, that the ecological condition of many WHS is in decline.

The work in **chapter 5** of this thesis informed the development of an IUCN technical report on "World Heritage, Wilderness, Large Landscapes and Seascapes", which provides pragmatic guidance to the World Heritage Committee and its partners on how to strengthen wilderness conservation under the Convention (Kormos et al. 2017). Importantly, this official guidance document further justifies calls for the Convention to acknowledge the contribution of wilderness areas to the outstanding universal value of WHS. Ideally modifications to the text of the Convention's Operational Guidelines will include the word wilderness in the Natural World Heritage criteria and/or conditions of integrity and management and protection. This may be a lengthy and challenging process, but modifications to the text have occurred previously in line with evolving conservation thinking. Such explicit acknowledgment of the importance of wilderness areas to humanity's collective heritage would further raise the profile of wilderness conservation worldwide, and possibly set a precedent for other conventions and multilateral environmental agreements such as the CBD to follow.

In **Chapter 6**, I analyzed patterns of forest loss in Niassa National Reserve in remote Northern Mozambique. This work was carried out in collaboration with The Wildlife Conservation Society, who are putting considerable effort and funding into protecting Niassa's wildlife and ecosystems. I found that Niassa lost >100km^2 of forest between 200 and 2012 amounting to 1% of its total forest extent. Although concerning, this loss was substantially lower than losses in the surrounding landscape suggesting that Niassa is performing well at protecting forest within its borders. Forest loss within Niassa was localized to main roads and large villages, which is important information the Niassa management team are using to target community engagement efforts and conservation action. The work in **chapter 6** directly supported the development of a ten-year management plan for Niassa, which will run until 2027. Niassa still contains large extents of forest, retaining much of it's wilderness value, and falls within a WHC coverage gap identified in chapter 5. As such, it potentially warrants World Heritage Status to increase the WHCs coverage of wilderness.

Implications for Future Conservation Strategy

Although the WHC is one of the most powerful international conservation instruments globally, and has the potential to make a major contribution to wilderness conservation, just retaining samples of wilderness areas within World Heritage sites will not be sufficient to secure all wilderness. I believe that the Earth's remaining wilderness can only be protected if their importance is recognised within multiple international policy frameworks aimed at conserving biodiversity, avoiding climate change, and achieving sustainable development, and if global targets are established for wilderness protection. Targets can be incorporated into existing policy frameworks immediately. For example, the carbon sequestration and storage capacities of wilderness areas could be written into the policy recommendations of the United Nations Framework Convention on Climate Change (UNFCCC) via the process for reducing emissions from forest loss or forest degradation (REDD+). To date, this process has focused on compensating landowners if they refrain from clearing an area of tropical forest that they had planned to develop. However, there is no incentive for nations, private industry and communities to protect carbon rich wilderness when no development is imminently planned. Therefore, there is nothing to stop the slow erosion of wilderness areas by small scale and unplanned industrial activity. New complementary approaches that reward the long-term maintenance of existing carbon stocks could be included, and similar policies are needed for carbon-rich wilderness beyond forests such as sea grass meadows, peat and temperate and boreal forests. Changes such as these would enable nations to make wilderness protection an integral part of their strategy for reducing emissions.

One potential avenue for improving wilderness protection could be to designate some of them as Key Biodiversity Areas (KBAs). The KBA approach identifies sites that "contribute significantly to the global persistence of biodiversity" based on a set of globally standardised and agreed criteria (IUCN 2016). Sites qualify if they meet one or more of the criteria, which fall under five categories; threatened biodiversity, geographically restricted biodiversity, ecological integrity, biological processes, and, irreplaceability. Wilderness areas align closely with all of these criteria making them excellent potential KBA candidates, although they are most closely aligned with the criterion on ecological integrity (Criterion C). To qualify under Criterion C, a site must be in excellent ecological condition so that it supports intact species assemblages and ecological processes in their natural state, with minimal human disturbance. This wording has specifically been included in the KBA standard to help protect wilderness areas. However, there are several challenges. Only two sites per ecoregion can qualify based on Criterion C, meaning that only a limited number, albeit a representative sample, of wilderness areas will be captured. Secondly, KBA status does not confer any formal protection for an area – they are conservation priorities that may warrant formal protection (Smith et al. 2019). Wilderness areas are increasingly acknowledged as conservation priorities in their own right (Watson et al. 2018b), but hopefully KBA status might facilitate them gaining formal protection.

Perhaps the most immediate opportunity to set a target for wilderness protection lies with the CBD. The CBD, is a platform that attempts to coordinate international action to halt or reverse biodiversity loss (CBD 2011), and is one of the most important international agreements relating to biodiversity conservation, with 196 countries currently working towards commitments outlined in in the 2020 Strategic Plan. These targets shape the behaviours of individuals, governments and non-government organisations, who appear to take their commitments seriously. This is demonstrated by the massive recent expansion of the global PA estate in response to Aichi target 11 (IUCN and UNEP-WCMC 2018), which mandates the inclusion of at least 17% of terrestrial areas and 10% of marine areas in effectively managed and ecologically representative protected areas by 2020 (CBD 2011). Despite progress towards these targets, biodiversity is still declining and habitat is being rapidly lost (Tittensor et al. 2014, Watson et al. 2016b). The CBD's next Strategic Plan for Biodiversity will take effect in 2020 and run until 2030, representing a crucial opportunity for the conservation community to learn from past mistakes and develop smarter, more ambitious targets (Maxwell et al. 2015a, Butchart et al. 2016). Looking across the chapters of this thesis highlights some important conclusions that are relevant for this process.

Bold yet achievable targets

One emergent theme from this thesis is the need to think about conservation and environmental management at a scale large enough to solve the environmental challenges we face in the Anthropocene. A step towards this could be to radically increase protection coverage targets (i.e. post Aichi Target 11) to a point where they are sufficient to achieve environmental outcomes such as averting the biodiversity crisis and ensuring wilderness areas remain intact (Di Marco et al. 2016b, Lovejoy 2017). Unfortunately, the current targets fall well short of this mark and require serious rethinking moving forward (Noss et al. 2012, Larsen et al. 2014). The need for conservation to think big has been championed by two recent initiatives, ‘half Earth and ‘nature needs half’, which boldly call for 50% of the planet to be conserved in an intact state (Wilson 2016, Dinerstein et al. 2017). The ethics and feasibility of these proposal have been subject to much debate (Büscher et al. 2016). However, there is a growing scientific consensus that coverage targets need to be substantially increased (Noss et al. 2012, Larsen et al. 2014, O'Leary et al. 2016, Dinerstein et al. 2017), although an exact number remains ambiguous. In the next section on ‘future research directions’ I discuss a possible methodology to answer this.

Beyond targets for the expansion of protected areas, a new target could also be established for the retention of natural ecosystems (Maron et al. 2018). The previous Strategic Plan for Biodiversity contained a habitat conversion target (Aichi Target 5) which stipulated that natural habitat loss should be halved, and “where feasible” eliminated. This target has many problems, firstly, there is no distinction between the value of intact or degraded natural habitat. Both are assumed to be equal but this thesis (particularly chapter 1) argues that this is clearly not the case – retaining intact habitats is far more important than stopping the loss of already degraded ones, and should be reflected in the target. Secondly, the target is insufficient and is so ambiguous that almost any action or outcome can be defended as success. This is particularly crucial for wilderness areas, where habitat loss in one part of the system can undermine the functioning of the whole (Laurance 2005, Peres 2005). For example, deforestation in the Amazon rainforest is reducing forest cover to the point where it is perilously close to a key threshold, below which the entire hydrological cycle that supports its humid forests could collapse (Lovejoy and Nobre 2018). The consequences of this would be devastating for people and nature, making it clear that we need to retain the vast majority of current wilderness extent to guarantee the persistence of the key planetary functions that support life on Earth (**chapter 1**). Halving the rate of habitat loss ‘where feasible’ will not suffice for wilderness areas, and revising this target in the next Strategic Plan for Biodiversity is a global imperative. I suggest a bold yet achievable target is to conserve 100% of Earth’s remaining intact ecosystems. In already degraded ecosystems the focus should be on preventing immediate extinctions and species declines, either via protection or restoration efforts, and in highly human dominated areas retaining human-nature interactions (Maron et al. 2018).

Linking Conservation Targets to Conservation Outcomes

Another common theme emerging from this thesis is the need to develop quantitative metrics to monitor conservation outcomes, and incorporate these into reporting towards international targets (Watson et al. 2016a, Barnes et al. 2018). For example, regarding Aichi Target 5, around 60% of national reports to the CBD indicate that progress on habitat loss is being made, but remotely sensed data show that forest loss exceeded regrowth in the vast majority of all nations between 2000-2012 (Hansen et al. 2013b). Similarly, **appendix 2** shows that 10% of wilderness was lost in just two decades (Watson et al. 2016c), and habitat loss outpaced protection efforts in two-thirds of countries globally (Watson et al. 2016b). Furthermore, human pressure increases and forest loss are widespread on protected land, preventing protected areas from achieve their potential to conserve biodiversity and wilderness (Allan et al. 2017b, Jones et al. 2018b). The current predominantly bottom-up ways nations report progress towards conservation targets is clearly failing to hold them accountable for the mismanagement of conservation areas and damage to the natural environment.

A potential solution to this problem is to officially incorporate quantitative metrics into reporting on progress towards conservation targets. **Chapters 2, 5 and 6** of this thesis demonstrate the utility of the wilderness maps, human footprint data and global forest watch data for quantitatively monitoring conservation assets. The finding of **chapter 5**, that many WHS were being highly damaged, was controversial given the implication that certain management authorities may have underperformed, and the inherently political nature of World Heritage. Remote sensing does have well discussed limitations (see discussion sections of **chapters 5-6** and (Hansen et al. 2013a, Achard et al. 2014, Tropek et al. 2014)), but satellites ultimately provide a real world picture of the state of the environment and thus go beyond politics and can help prevent nations from downplaying or omitting politically unfavourable results. Given the political nature of land conversion globally (Burgess et al. 2012, Reside et al. 2017), incorporating quantitative metrics into reporting towards conservation targets, such as Aichi Targets 5 and 17, would constitute a major advance for global conservation (Jones et al. 2018b). The WHC is particularly well placed to take the lead, and the IUCN could directly incorporate the analyses in **chapter 4** into their World Heritage Outlook Reports, which come out every 2-3 years, and are still based predominantly on qualitative reporting methods (Osipova et al. 2017).

Future Research Directions

In the previous sections, I discussed some of the themes and conclusions that emerge from this thesis as a whole. These included the need to plan for conservation and environmental stewardship at a planetary scale, and the need to improve the management of conservation assets such as protected areas. Each of these represents an important avenue for future research, which I explore further in this section.

Sufficient Targets for Nature Conservation

The question 'how much do we need to conserve?' is fundamentally important for developing conservation targets that are sufficient to address the biodiversity crisis, and identifying priority areas for conservation action. This action could occur in multiple ways, for example, through targeting protected areas, restoring degraded habitat, or retaining intact habitat. I believe there is significant value to a research agenda aiming to determine how much land is required to secure nature and its values, since this will help provide an ecological basis for the development of future conservation targets.

Developing a global conservation plan would require accounting for a diverse range of conservation and environmental objectives, and building consensus on these objectives will not be a simple task. However, a first step could be to include current conservation features such as the global protected area estate. This could be complemented by identifying places that hold the last populations of a species or an ecosystem, for example Key Biodiversity Areas (IUCN 2016), and Alliance for Zero Extinction sites (AZE 2010), since securing them is essential to avoid immediate extinctions. These features could be complemented further by including large ecologically intact ecosystems that still function in a predominantly natural state, such as wilderness areas, which are essential for securing a range of environmental and cultural values.

Combined, the above datasets could form the backbone of a global plan for nature, which could extend across the marine, terrestrial and freshwater realms. By carrying out a global gap analysis assessing species coverage, and then a prioritisation analysis to ensure an adequate proportion of all species ranges is protected, we could determine an aerial percentage of how much of Earth is sufficient to protect nature. These prioritisation analyses could be done using state of the art conservation planning tools such as Marxan or integer linear programming (Watts et al. 2009, Beyer et al. 2016, Hanson et al. 2018). This is just one method of developing a spatially explicit global plan for nature but I am confident it would provide useful information to support the development of new targets in the CBD's updated Strategic Plan for biodiversity.

Forward looking Conservation: Modelling Future Risks

Threats to the environment are predicted to increase substantially in the near future, as human populations and consumption levels rise (Crist et al. 2017, Tilman et al. 2017). Forecasting the scope and intensity of these threats is essential information for ensuring conservation actions are targeted to at risk places to avert impacts such as species declines or wilderness loss. This is especially important for wilderness areas, which cover roughly 30 million km^2, so to conserve them within protected areas would require a near doubling of the current protected area estate which is unlikely. It would be much more efficient to target protection, or other conservation actions, to sites that are most at risk (Pressey et al. 2017, Venter et al. 2017). However, identifying these places is challenging because it requires estimating future risk, which involves complex and uncertain modelling techniques.

I anticipate that efforts to develop spatially explicit projections of threats will become a major front for conservation research, especially as access to big data and supercomputing continue to advance. Future research could aim to identify which wilderness areas are at the greatest risk of future decline. Wilderness loss from 1993-2009 could be modelled using multiple data sources (drivers), such as distance from human influence, agricultural value and suitability of land, elevation and slope of land, and existence of known petroleum reserves, to derive a statistical relationship for each driver in a given region (because not all drivers will be significant everywhere). It would then be possible to identify where future loss is likely to occur, and predict the area of future wilderness loss by assuming the future rate will be the same as the past and forecasting within each region. Using an ensemble approach that combines multiple wilderness loss scenarios to generate final forecast probabilities of loss would help account for the uncertainty associated with predicting future land-use change

Systematically Upgrading Protected Areas

The recent expansion of the global protected area estate is one of conservations greatest success stories. The global extent of protected land has doubled since the 1992 Earth summit in Rio de Janeiro, Brazil, to include over 200,000 protected areas covering 14.7% of Earth's terrestrial surface and 7.3% of our oceans (IUCN and UNEP-WCMC 2018). However, it appears the protected area estate currently exceeds the conservation community's capacity to adequately fund and manage it (Gill et al. 2017, Jones et al. 2018b).

Furthermore, many protected areas have high levels of human pressure within their boundaries, as demonstrated in **chapters 5-6**, compromising their ability to meet both biodiversity and broader societal objectives (Allan et al. 2017a, Jones et al. 2018b).

Recognizing this problem, there have been recent calls for increased investment in upgrading protected areas that have high, but currently un-realized potential for both biodiversity conservation and human communities (Pringle 2017). By upgrading protected areas, I mean increasing management effectiveness, while harmonizing them with the needs and aspirations of their constituencies (Pringle 2017). Organizations such as African Parks (African Parks 2017), and the Tompkins Foundation (Tompkins Conservation 2016) are already investing in upgrading protected areas, and setting a global precedent. However, it appears that protected areas have been selected for upgrading based primarily on factors such as political will, charismatic species or spectacular landscapes. Although these factors are important, conservation gains could be increased by including some of the principles of conservation planning (Margules and Pressey 2000), for example identifying protected areas for upgrading that increase the representation of species inside well managed reserves.

In the next few years, I plan to develop a structured approach for identifying priority-protected areas for upgrading. In its simplest form, this could involve first identifying protected areas that are currently well managed and analyzing their contribution to conservation targets (e.g. adequately representing species). Building off this baseline, I will then identify the best underperforming protected area to upgrade next, based on its additional contribution to biodiversity conservation and broader societal objectives, and the degree to which it is threatened. This could be repeated iteratively using a new baseline of well-managed protected areas each time to rank protected areas in order of priority. Additional complexity could be added by including estimates of intervention cost and likelihood of success. I will seek partnerships and collaboration with the organizations actively involved in upgrading protected areas.

Concluding remarks

Human pressure on planet earth will only increase as our population climbs and technology continues its relentless advance. Our window of opportunity to safeguard human well-being and the health of our planet is closing fast. Securing wilderness is an essential step towards this, as these areas house exceptional biodiversity, cultural, and climate regulation values.

Wilderness values can only be conserved in their current, naturally functioning state. Once they are degraded, full restoration is near impossible. As President Lyndon B. Johnson observed when he signed the United States Wilderness Act in 1964 “If future generations are to remember us with gratitude rather than contempt, we must leave them something more than the miracles of technology. We must leave them a glimpse of the world as it was in the beginning, not just after we got through with it”. “Once our natural splendour is destroyed, it can never be recaptured. And once man can no longer walk with beauty or wonder at nature, his spirit will wither and his sustenance be wasted”. Given we have already lost so much, we must grasp this opportunity to secure wilderness before it disappears forever.

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APPENDIX 1 Global terrestrial Human Footprint maps for 1993 and 2009

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Abstract

Remotely-sensed and bottom-up survey information were compiled on eight variables measuring the direct and indirect human pressures on the environment globally in 1993 and 2009. This represents not only the most current information of its type, but also the first temporally-consistent set of Human Footprint maps. Data on human pressures were acquired or developed for: 1) built environments, 2) population density, 3) electric infrastructure, 4) crop lands, 5) pasture lands, 6) roads, 7) railways, and 8) navigable waterways. Pressures were then overlaid to create the standardized Human Footprint maps for all non-Antarctic land areas. A validation analysis using scored pressures from 3114 x 1 km^2 random sample plots revealed strong agreement with the Human Footprint maps. We anticipate that the Human Footprint maps will find a range of uses as proxies for human disturbance of natural systems. The updated maps should provide an increased understanding of the human pressures that drive macro-ecological patterns, as well as for tracking environmental change and informing conservation science and application.

Background & Summary

Human pressures on the environment are the actions taken by humans with the potential to harm nature (Borja et al. 2006, Martins et al. 2012). Cumulative pressure mapping measures the breadth of these pressures by coupling top-down remote sensing of land cover change with data on additional human pressures collected 'bottom-up' through systematic surveys and modelling (Vorosmarty et al. 2010, Halpern and Fujita 2013). The method circumvents the limitations of using remote sensing alone, which has difficulty in detecting low intensity pressures (Potapov et al. 2008), such as linear infrastructures (Laurance et al. 2009) and pasture lands (Wassenaar et al. 2007), and often confounds natural and anthropogenic land covers in arid and mosaic environments (Herold et al. 2008).

Cumulative pressure maps have been developed at regional (Woolmer et al. 2008, Maxwell et al. 2013) and global scales(Halpern et al. 2008, Geldmann et al. 2014). The 'Human Footprint' was first released in 2002 using data primarily from the early 1990s (approximately 1993) on eight human pressures globally, making it the most complete, highest resolution and globally-consistent terrestrial dataset on cumulative human pressures on the environment (Sanderson et al. 2002). It has been used in a large number of ecological and conservation analyses, and still receives around 100 citations each year, particularly from its data users. However, the Human Footprint is a static and dated view of human pressures on the environment. With many of Earth's systems experiencing pressures close to or beyond safe levels (Steffen et al. 2015b), there is a strong need for an up-to-date understanding of the spatial and temporal trends in human pressures.

Here we use the Human Footprint methodology (Sanderson et al. 2002) to compile remotely-sensed and bottom-up survey information on eight variables measuring the direct and indirect human pressures on the environment in 1993 and 2009. This synthesis represents not only the most current information of its type, but also the first temporally-consistent set of Human Footprint maps, allowing for analyses of change over time. We also provide the first validation of a cumulative pressure map by adopting methods from remote sensing (Congalton 2001) to visually interpret human pressures in high resolution (median = 0.5 m) imagery from 3114 1 km^2 random sample plots globally. We then determine the level of agreement between these visually interpreted pressures and those mapped by the Human Footprint.

The Human footprint maps provide information on where humans are exerting pressure on natural systems, altering them from their natural states. They also provide information on where these pressures are absent, and ecosystems are likely to be operating in a more natural state. These pressure-free lands represent candidate sites for consideration as 'Wilderness' (Mittermeier et al. 2003, Watson et al. 2009). The new Human Footprint maps have already been used to show that recent economic and population growth has far outstripped increases in the Human Footprint, yet the most biologically diverse regions of Earth have been disproportionately impacted (Venter et al. 2016c). We anticipate that the 1993 and 2009 Human Footprint maps will find a range of additional uses, such as serving as proxies for human disturbance and wilderness, including understanding the role of human pressures in driving macro-ecological patterns (Mayor et al. 2012, Seiferling et al. 2014), species extinction risk and distribution analyses (Di Marco and Santini 2015), dispersal ecology (Hand et al. 2014), conservation science and decision making (Tulloch et al. 2015b), and tracking progress toward policy commitments to conservation (Büscher et al. 2016), among others.

Methods

Overview of methods for mapping the Human Footprint

To create the Human Footprint maps we adopted the methods developed by Sanderson and colleagues (2002). Data on human pressures in 1993 and 2009 were collected or developed for: 1) the extent of built environments, 2) population density, 3) electric infrastructure, 4) crop lands, 5) pasture lands, 6) roads, 7) railways, and 8) navigable waterways, which are described in detail below (Figure A1.1 step 1). To facilitate comparison across pressures we placed each human pressure within a 0 – 10 scale (Figure A1.1 step 2), weighted within that range according to estimates of their relative levels of human pressure following Sanderson et al. (2002). The resulting standardized pressures were then summed together to create the standardized Human Footprint maps for all non-Antarctic land areas (Figure A1.1 step 3). Pressures are not intended to be mutually exclusive, and many will co-occur in the same location. Three pressures only had data from a single time period, and these are treated as static in the Human Footprint maps.

We used ArcGIS 10.1 to integrate spatial data on human pressures. Analyses were conducted in Mollowedie equal area projection at the 1km^2 resolution, yielding ~134.1 million pixels for Earth's non-Antarctic terrestrial surface. For any grid cell, the Human Footprint can range between 0 – 50. The following sections and Table A1.1 describe in detail the source data for each pressure, the processing steps applied, and the rationale behind the pressure weighting, and the output datasets created.

Figure A1.1 Workflow of the Human Footprint approach to mapping cumulative human pressures on the environment.

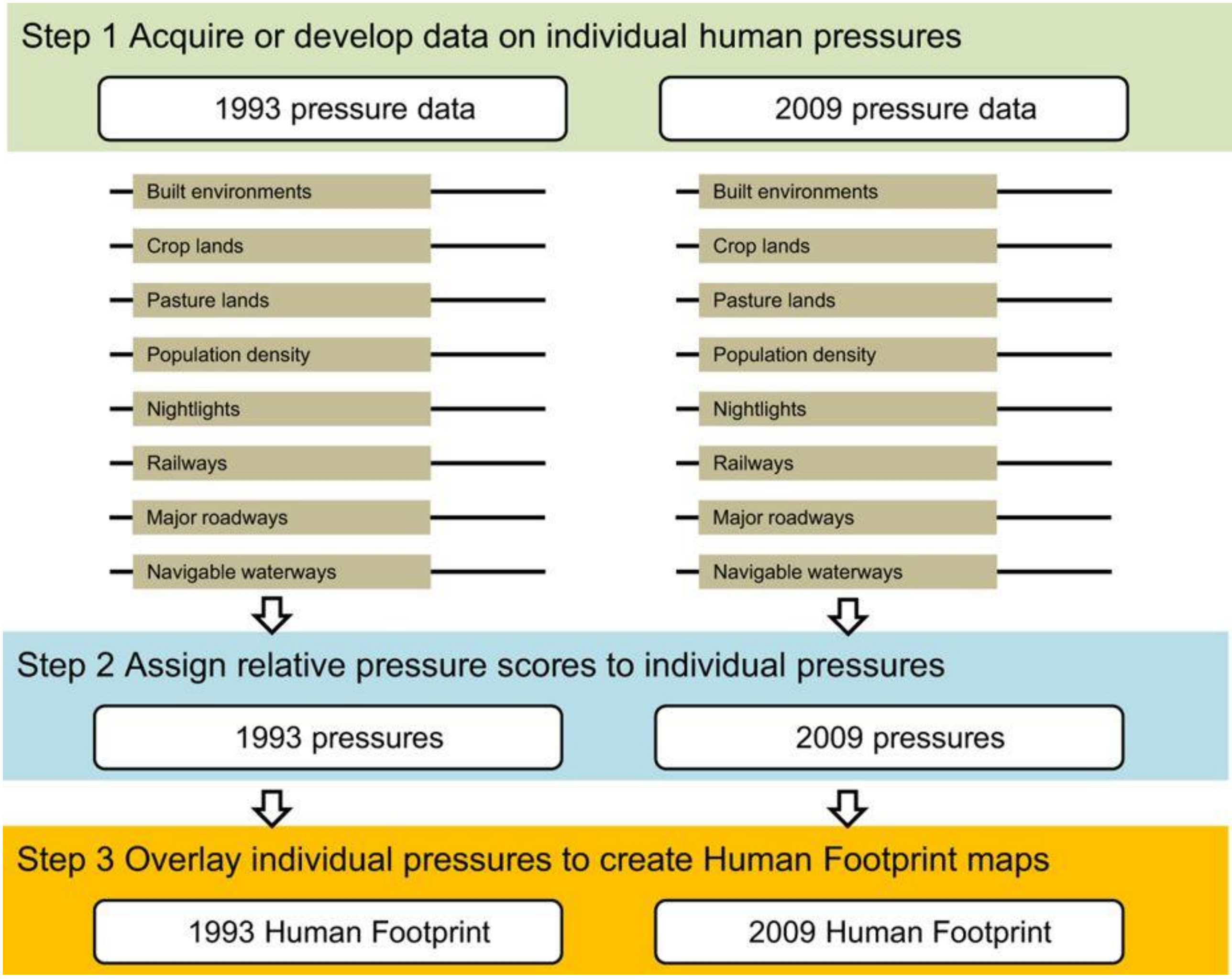


Table A1.1 Summary of data inputs, manipulations and outputs in the Human Footprint workflow.

Source	Data used	Temporal range	Resolution	Data manipulations	Outputs
REF 27	Average, stable lights, & cloud free coverages	1994, 2009	30 arc second, ~1 km at equator	1) Intercalibrate across years 2) reproject and resample to 1 km raster basemap	Built1994.tif Built2009.tif

				3) Convert to binary map of areas exhibiting a Digital Number equal to or above ‘20’. 4) Assign these areas the pressure score of ‘10’.	
Data Citation 2	Gridded population of the world GPWv3, density grids	1990, 2010	2.5 arc minute, ~5 km at equator	1) reproject and resample to 1 km raster basemap 2) Assign pressure score using eq. 1 in methods	Popdensity1190.tif Popdensity2010.tif
REF 27	Average, stable lights, & cloud free coverages	1994, 2009	30 arc second, ~1 km at equator	1) Intercalibrate across years 2) reproject and resample to 1 km raster basemap 3) Create 11 equal quintile bins for 1994. 4) Assign pressure scores to bins from 0 – 10, for 1994, and using the same DN thresholds for 2009.	Lights1994.tif Lights2009.tiff
Data Citation 3	University of Maryland Global Land Cover Classifications 1992-1993	1992 - 1993	1 km	1) reproject and resample to 1 km raster basemap 2) Convert to binary map showing crop lands 3) Exclude all areas already mapped as built 4) Assign crop lands a	Croplands 1993.tif

				pressure score of ‘7’.	
REF 40	GlobCover Version 2.3 2009	2005 - 2006	300m	1) reproject and resample to 1 km raster basemap 2) Convert to binary map showing crop lands 3) Exclude all areas already mapped as built 4) Assign crop lands a pressure score of ‘7’.	Croplands 2005.tif
REF 41	M3-Pasture data	2000	5 minute, ~10 km at equator	1) reproject and resample to 1 km raster basemap 2) Exclude all areas already mapped as built or crop lands 3) Assign pressure score of 4, weighted by percent pasture lands	Pasturelands.tif
Data Citation 4	Global Roads Open Access Data Set (gROADS) v1	1980 - 2010	Vector data, accurate to 500m	1) reproject and covert to 1 km raster basemap 2) Exclude trails and private roads 3) Assign pressure score of ‘8’ to roaded pixels, and ‘4’ to adjacent pixels, exponentially decaying to 0 at 15km.	Roadways.tif
REF 30	Vector Map Level 0 (VMap), railways	~1990	Vector data, accurate to 1 km	1) reproject and covert to 1 km raster basemap	Railways.tif

				2) Assign pressure score of ‘8’ to rail pixels	
REF 48	HydroSHEDS, stream discharge	No timeframe	3 arc second, ~100 m at equator	1) reproject and covert to 1 km raster basemap 2) Use eq 2-5 to determine stream depth 3) Exclude all stream reaches less than 2m 4) exclude all reaches no within 80 km of a stream bank which is within 4km of a pixel with a DN > 4, in 1994 or 2009. 5) Add coastlines within 80 km of a coastal bank which is within 4km of a pixel with a DN > 4, in 1994 or 2009. 6) Assign pressure score of ‘4’ to adjacent pixels, exponentially decaying to 0 at 15km.	Navwater 1994.tif Navwater 2009.tif

Built environments

Built environments are human produced areas that provide the setting for human activity. In the context of the human footprint, we take these areas to be primarily urban settings, including buildings, paved land and urban parks. Built environments do not provide viable habitats for many species of conservation concern, nor do they provide high levels of ecosystem services (Tratalos et al. 2007, Chamberlain et al. 2009, Butchart et al. 2010, Aronson et al. 2014). As such, built environments were assigned a pressure score of 10.

To map built environments, we used the Defence Meteorological Satellite Program Operational Line Scanner (DMSP-OLS) composite images which gives the annual average brightness of 30 arc second (~1 km at the equator) pixels in units of digital numbers (DN) (Elvidge et al. 2001). These data are provided for each year from 1992 to 2012. We extracted data for the years 1994 (1993 was excluded due to anomalies in the data), and 2009, and both datasets were then inter-calibrated to facilitate comparison (Elvidge et al. 2009). Using the DMSP-OLS datasets, we considered pixels to be 'built' if they exhibited a calibrated DN greater than 20. We selected this threshold based on a global analyses of the implications of a range of thresholds for mapped extent of cities (Small et al. 2011), and visual validation against Landsat imagery for 10 cities spread globally.

The DMSP-OLS has limitations for the purpose of mapping human settlements, including hyper sensitivity of the sensors causing detection of over-glow adjacent to built environments (Small et al. 2011) and bright lights associated with gas flaring from oil production facilities (Elvidge et al. 2009). However, no other data exist to map built environments in a consistent way globally over our time horizon. While other datasets provide a one-year snap shot of urban extent, they cannot be compared across time due to large differences in the methodologies used (NIMA 1997, CIESEN 2005a, Schneider et al. 2009), and the wildly contrasting extents in mapped built environments.

Population density

Many of the pressures humans impose on the environment are proximate to their location, such as human disturbance, hunting and the persecution of non-desired species (Brashares et al. 2001). Moreover, even low-density human populations with limited technology and infrastructure developments can have significant impacts on biodiversity, as evidenced by the widespread loss of various taxa, particularly mega fauna, following human colonization of previously unpopulated areas (Burney and Flannery 2005, Miller et al. 2005).

Human population density was mapped using the Gridded Population of the World dataset developed by the Centre for International Earth Science Information Network (CIESEN) (CIESEN 2005b). The dataset provides a ~4km x ~4km gridded summary of population census data for the years 1990 and 2010, which we downscaled using bilinear sampling in ArcGIS 10.1 to match the 1km^2 resolution of the other datasets. For all locations with more than 1000 people/km^2, we assigned a pressure score of 10 (Table A2.2). For more sparsely populated areas with densities lower than 1000 people/km^2, we logarithmically scaled the pressure score using,

$$\text{Pressure score} = 3.333 * \log(\text{population density} + 1) \qquad (1)$$

Human population density is scored in this way under the assumption that the pressures people induce on their local natural systems increase logarithmically with increasing population density, and saturate at a level of 1000 people per km^2.

Night-time lights

The high sensitivity of the DMSP-OLS (Elvidge et al. 2001) dataset provides a means for mapping the sparser electric infrastructure typical of more rural and suburban areas. In 2009, 79% of the lights registered in the DMSP-OLS dataset had a Digital Number less than 20, and are therefore not included in our ‘built environments’ layers. However, these lower DN values are often important human infrastructures, such as rural housing or working landscapes, with associated pressures on natural environments.

To include these pressures, we used the inter-calibrated DMSP-OLS layers (Elvidge et al. 2001) used for the built environments mapping. The equations for intercalibrating across years are second order quadratics trained using data from Sicily, which was chosen as it had negligible infrastructure change over this period and where DN average roughly 14 (Elvidge et al. 2001). For our purposes, DN values of six or less where excluded from consideration prior to calibration of data, as the shape of the quadratic function leads to severe distortion of very low DN values. The inter-calibrated DN data from 1994 were then rescaled using an equal quintile approach into a 0-10 scale (Table A1.2). To scale the data, we divided the calibrated night light data into 10 equal sample bins (each bin with a DN greater than 1 contains the same number of pixels) based on the DN values and then assigned them scores of 1 through 10, starting with the lowest DN bin. DN values of 0 were assigned a score of 0. The thresholds used to bin the 1994 data where then used to convert the 2009 data into a comparable 0-10 scale.

Table A1.2 Pressure scheme used to assign weights to the eight individual pressures in the Human Footprint maps.

Pressure	**Score**	**Details**
Built environments	0,10	All areas mapped as build given score of 10.

Population density	0-10 continuous	Pressure score = 3.333 * log (population density + 1)
Night-time lights	0-10 continuous	Equal quintile bins
Croplands	0,7	All areas mapped as crops given score of 7.
Pasture	0,4	All areas mapped as pasture given score of 4.
Roads	0,8 Direct impacts 0-4 indirect impacts	500m either side of roads given a direct pressure score of 8 Starting 500m out from road, pressure score of 4 exponentially decaying out to 15km.
Railways	0,8	500m either side of railways given a direct pressure score of 8 Starting 500m out from road
Navigable waterways	0-4	pressure score of 4 exponentially decaying out to 15km.

Crop and pasture lands

Crop lands vary in their structure from intensely managed monocultures receiving high inputs of pesticides and fertilizers, to mosaic agricultures such as slash and burn methods that can support intermediate levels of natural values (Luck and Daily 2003, Fischer et al. 2008). For the purposes of the human footprint, we focused only on intensive agriculture because of its greater direct pressure on the environment, as well as to circumvent the shortcomings of using remotely sensed data to map mosaic agriculture globally, namely the tendency to confound agriculture mosaics with natural woodland and savannah ecosystems (Herold et al. 2008).

Spatial data on remotely sensed agriculture extent in 1992 were extracted from the UMD Land Cover Classification (Hansen et al. 2000), and for 2009 from GlobCover (ESA 2011).

Although intensive agriculture often results in whole-scale ecosystem conversion, we gave it a pressure score of 7 (Table A1.2), which is lower than built environments because of their less impervious cover.

Pasture lands cover 22% of the Earth’s land base or almost twice that of agricultural crops (Ramankutty et al. 2008), making them the most extensive direct human pressure on the environment. Land grazed by domesticated herbivores is often degraded through a combination of fencing, intensive browsing, soil compaction, invasive grasses and other species, and altered fire regimes (Kauffman and Krueger 1984). We mapped grazing lands for the year 2000 using a spatial dataset that combines agricultural census data with satellite derived land cover to map pasture extent (Ramankutty et al. 2008). We assigned pasture a pressure score of 4, which was then scaled from 0 – 4 using the percent pasture for each $1km^2$ pixel.

Roads and railways

As one of humanity’s most prolific linear infrastructures, roads are an important direct driver of habitat conversion (Trombulak and Frissell 2000). Beyond simply reducing the extent of suitable habitat, roads can act as population sinks for many species through traffic induced mortality (Woodroffe and Ginsberg 1998). Roads also fragment otherwise contiguous blocks of habitat, and create edge effects such as reduced humidity (Laurance et al. 2009) and increased fire frequency that reach well beyond the roads immediate footprint (Adeney et al. 2009). Finally, roads provide conduits for humans to access nature, bringing hunters and nature users into otherwise wilderness locations (Forman and Alexander 1998).

We acquired data on the distribution of roads from gROADS (CIESEN 2013), and excluded all trails and private roads, which were inconsistently mapped, with only a subset of countries mapping their linear infrastructure to this resolution. The dataset is the most comprehensive publicly available database on roads, which compiles nationally mapped road data spanning the period 1980-2000 and has a spatial accuracy of around 500m. The gROADS data do not include all minor roads, and therefore should be viewed as a map of the major roadways. We mapped the direct and indirect influence of roads by assigning a pressure score of 8 for 0.5 km out for either side of roads, and access pressures were awarded a score of 4 at 0.5 km and decaying exponentially out to 15 km either side of the road (Table A1.2).

While railways are an important component of our global transport system, their pressure on the environment differs in nature from that of our road networks. By modifying a linear swath of habitat, railways exert direct pressure where they are constructed, similar to roads. However, as passengers seldom disembark from trains in places other than rail stations, railways do not provide a means of accessing the natural environments along their borders. To map railways we used the same dataset as was used in the original footprint (NIMA 1997), as no update of this dataset or alternate source has been developed. The direct pressure of railways where assigned a pressure score of 8 for a distance of 0.5 km on either side of the railway.

Navigable waterways

Like roads, coastlines and navigable rivers act as conduits for people to access nature. While all coastlines are theoretically navigable, for the purposes of the human footprint we only considered coasts (NIMA 1997) as navigable for 80 km either direction of signs of a human settlement, which were mapped as a night lights signal with a DN (Elvidge et al. 2001) greater than 6 within 4 km of the coast. We chose 80 km as an approximation of the distance a vessel can travel and return during daylight hours. As new settlements can arise to make new sections of coast navigable, coastal layers were generated for the years 1994 and 2009.

Large lakes can act essentially as inland seas, with their coasts frequently plied by trade and harvest vessels. Based on their size and visually identified shipping traffic and shore side settlements, we treated the great lakes of North America, Lake Nicaragua, Lake Titicaca in South America, Lakes Onega and Peipus in Russia, Lakes Balkash and Issyk Kul in Kazakhstan, and Lakes Victoria, Tanganyika and Malawi in Africa as we did navigable marine coasts.

Rivers were considered as navigable if their depth was greater than 2m and there were signs of nighttime lights (DN >= 6) within 4km of their banks, or if contiguous with a navigable coast or large inland lake, and then for a distance of 80 km or until stream depth is likely to prevent boat traffic (Table A1.2). To map rivers and their depth we used the hydrosheds (hydrological data and maps based on shuttle elevation derivatives at multiple scales) (Lehner et al. 2008) dataset on stream discharge, and the following formulae from (ESRI World Imagery , Bjerklie et al. 2003):

$$\text{Stream width} = 8.1 \times (\text{discharge } [m^3/s])^{0.58} \quad (2)$$

and,

$$\text{velocity} = 4.0 \times (\text{discharge } [m^3/s])^{0.6} / (\text{width } [m]). \quad (3)$$

and,

$$\text{Cross-sectional area} = \text{discharge}/\text{velocity} \quad (4)$$

and,

$$\text{depth} = 1.5 * \text{area} / \text{width} \quad (5)$$

Assuming second order parabola as channel shape.

Navigable rivers layers were created for the years 1994 and 2009, and combined with the navigable coasts and inland seas layers to create the final navigable waterways layers. The access pressure from navigable water bodies were awarded a score of 4 adjacent to the water body, decaying exponentially out to 15km.

Data Records

The 1 km^2 resolution, temporally-comparable Human Footprint maps [Data Citation 1] are stored in the Dryad Digital Repository, and may also be freely accessed through the Socioeconomic Data and Applications Center website (www.worldpop.org/data/). From Dryad the files may be downloaded as a single 7-zip file archive (7-Zip.org) which contains individual GeoTIFF datasets, an excel file with the validation data and a PDF with the validation key. The GeoTIFFs include the Human Footprint maps for 1993 and 2009 (Figure A1.2), as well 14 additional GeoTIFFs of the processed data for each of the eight pressures (Figure A1.1 step 2) from each time step (Table A1.1 and Table A1.3). The individual pressure layers are provided should data users wish to rework these data to create alternate maps of human pressure for their particular needs or region.

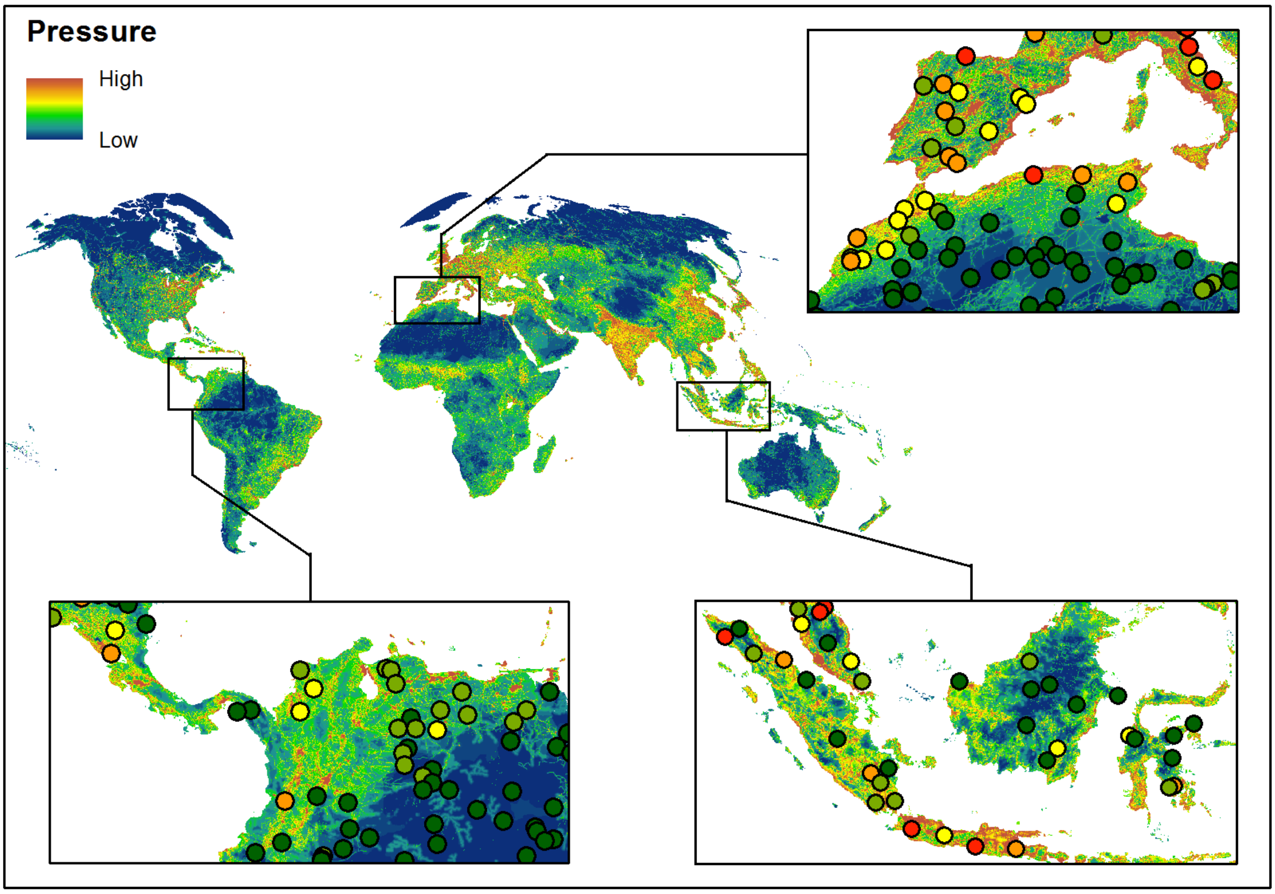
Table A1.3 The name, description and type of data included in the HumanFooptrint.zip file.

Name	Description	Format
Validation.xlsx	Data on visual score of pressures for 3114 1km^2 plots using high resolution imagery.	Excel

Appendix1.pdf	Key used to visually interpret human pressures.	PDF
HFP1993.tif	The Human Footprint camp of cumulative pressures on the environment in 1993.	GeoTIFF
HFP2009.tif	The Human Footprint camp of cumulative pressures on the environment in 2009.	GeoTIFF
Built1994.tif	Individual pressure map of built environments in 1994.	GeoTIFF
Built2009.tif	Individual pressure map of built environments in 2009.	GeoTIFF
Croplands1992.tif	Individual pressure map of crop lands in 1992.	GeoTIFF
Croplands2005.tif	Individual pressure map of crop lands in 2005.	GeoTIFF
Lights1994.tif	Individual pressure map of night-time lights in 1994.	GeoTIFF
Lights2009.tif	Individual pressure map of night-time lights in 2009.	GeoTIFF
Navwater1994.tif	Individual pressure map of navigable waterways in 1994.	GeoTIFF
Navwater2009.tif	Individual pressure map of navigable waterways in 2009.	GeoTIFF
Pasture1993.tif	Individual pressure map of pasture lands in 1993.	GeoTIFF
Pasture2009.tif	Individual pressure map of pasture lands in 2009.	GeoTIFF
Popdensity1990.tif	Individual pressure map of human population density in 1990.	GeoTIFF
Popdensity2010.tif	Individual pressure map of human population density in 2010.	GeoTIFF
Railways.tif	Individual pressure map of railways circa 1990.	GeoTIFF

Roads.tif	Individual pressure map of roads circa 2000.	GeoTIFF

Figure A1.2 The Human Footprint map for 2009, with panels showing regional overlays with the results of the validation plots.



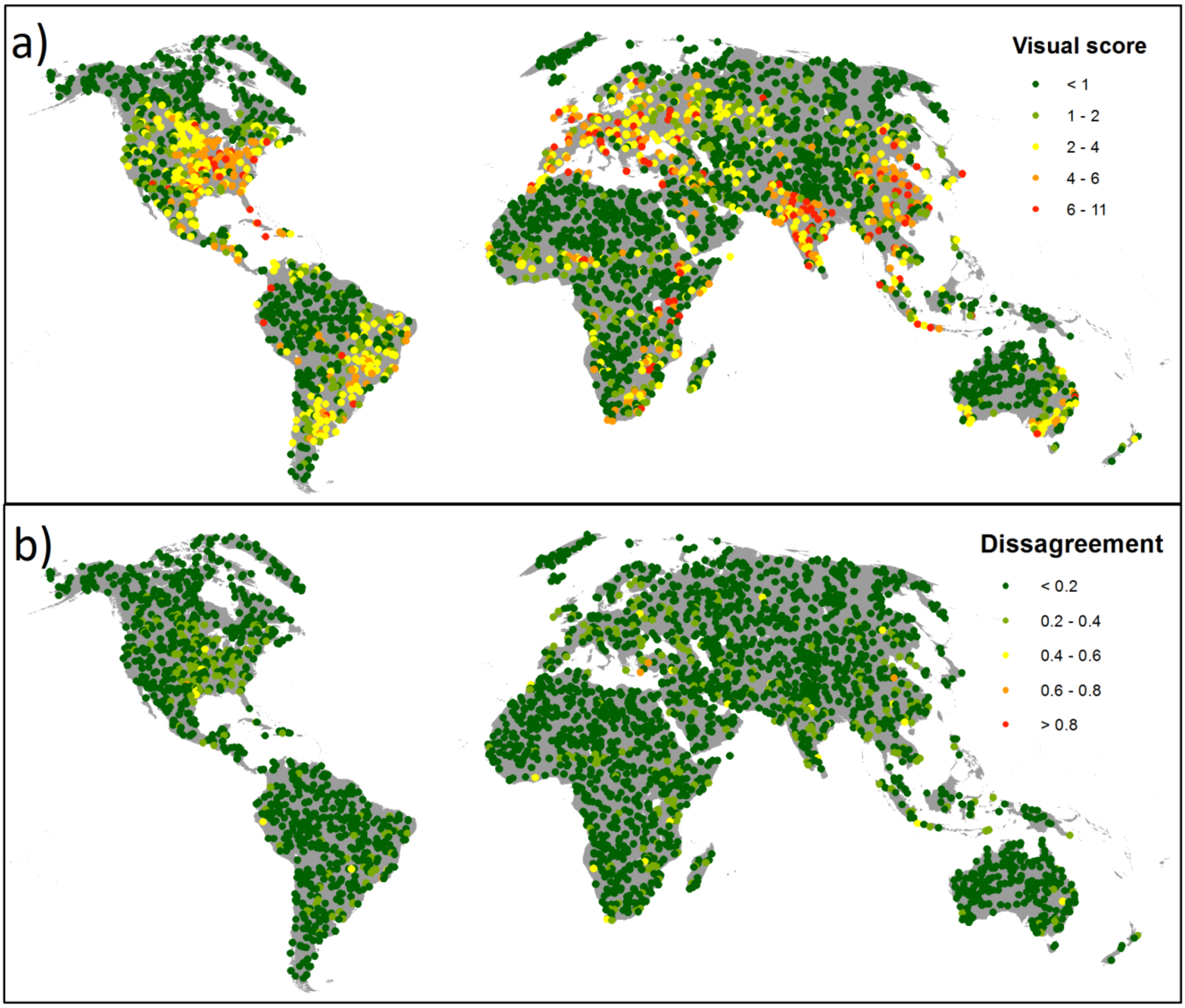
Technical Validation

High resolution images were used to visually interpret human pressures in 3460 x 1 km^2 sample plots randomly located across the Earth's non-Antarctic land areas (Figure A1.3a). Images for these plots were obtained from World Imagery (ESRI World Imagery), which provides one meter or better satellite and aerial imagery in many parts of the world and lower resolution satellite imagery worldwide. The map features 0.3m resolution imagery across the continental United States and parts of Western Europe, as well as many parts of the world, with concentrations in South America, Eastern Europe, India, Japan, the Middle East and Northern Africa, Southern Africa, Australia, and New Zealand. The imagery used for the validation plots had a median resolution of 0.5 meters and a median acquisition year of 2010. Comparable imagery was not available for the 1993 time period, and therefore only the 2009 map underwent validation.

For the visual interpretation, the extent of built environments, crop lands, pasture lands, roads, human settlements, infrastructures and navigable waterways, were recorded using a standard key for identifying these features. Shape, size, texture and colour of features in the imagery were important characteristics for identifying human pressures on the environment. Interpretations were also marked as ‘certain’ or ‘uncertain’, and the year and resolution of the interpreted image was recorded. The 346 ‘uncertain’ plots were discarded, leaving 3114 validation plots (Figure A1.3a). In general, plots were classified as ‘uncertain’ for two reasons; either because cloud cover obscured the image, or because only medium resolution (15m) imagery was available for the plot, preventing accurate interpretation of the image. The human footprint score for each plot was determined in ArcGIS, and the visual and Human Footprint scores were then normalized to a 0-1 scale. As we only retained plots for which visual interpretations of the images were determined to be ‘certain’, we consider the visual score to be the true state of in-situ pressures for the plots.

Two statistics were used to determine Human Footprint performance, root mean squared error (RMSE) (Cort and Kenji 2005) and the Cohen kappa statistic of agreement (Viera and Garrett 2005). The RMSE is a dimensioned (expresses average error in the units of variable of interest) error metric for numerical predictions, and tends to heavily punish large errors. The Kappa statistic expresses the agreement between two categorical datasets corrected for the expected agreement, which is based on a random allocation given the relative class sizes. When calculating the kappa statistic, the Human Footprint score was considered as a match to the visual score if they were within 20% of one another on the 0-1 scale.

Figure A1.3 Results from 3114×1 km^2 validation plots interpreted and scored



There is strong agreement between the Human Footprint measure of pressure and pressures scored by visual interpretation of high-resolution imagery. The RMSE for the 3114 validation plots was 0.125 on the normalized 0-1 scale, indicating an average error of approximately 13%. The Kappa statistic was 0.737 ($p < 0.01$), also indicating good agreement between the Human Footprint and the validation dataset. Of the 3114 1km^2 validation plots, the Human Footprint scored 94 of them 20% higher than the visual score and 263 of them 20% lower. The remaining 2757 plots (88.5%) were within 20% agreement. While this represents good agreement, it appears that the Human Footprint is to some extent susceptible to mapping pressures as absent in locations where they are actually present. The maps should therefore be considered as conservative estimates of human pressures on the environment. The Kappa statistic measure of agreement is sensitive to the threshold used to consider plots as a 'match'. If we apply a more stringent threshold for agreement of within 15% of one another, the Kappa statistic falls to 0.565 (moderate agreement), and if we apply a less stringent threshold of within 25%, the Kappa statistic increases to 0.856 (very high agreement).

While agreement is generally strong, there is some geographic variation in the RMSE results comparing the Human Footprint scores and those derived from visual interpretation (Figure A1.3b). By calculating RMSE for all biomes that contain at least 100 of the 3114 sample plots, we found that agreement was strongest in the Tundra biome and the Temperate grasslands, savannas and shrublands biomes (Table A1.4). Agreement was weakest in the Temperate broadleaf and mixed forest biome and the Boreal.

Table A1.4 Root Mean Square Errors results comparing the Human Footprint scores with 3114 validation plots globally, and for biomes with at least 100 plots within them.

Region	RMSE
RMSE Global	0.125706
RMSE Boreal	0.164053
RMSE Deserts and xeric shrublands	0.091757
RMSE Montane grasslands	0.121541
RMSE Temperate broadleaf and mixed forests	0.175661
RMSE Temperate grasslands, savannas, and shrublands	0.085226
RMSE Tropical and subtropical grasslands, savannas, and shrublands	0.121362
RMSE Tropical and subtropical moist broadleaf forests	0.142398
RMSE Tundra	0.028995

Usage Notes

Mapping human pressures to the environment is an essential first step to identifying priority areas for protection or restoration of natural systems. Understanding the spatial distribution of pressures, as well as their change through time, also provides insights for studies on macro-ecological patterns. The Human Footprint maps for 1993 and 2009 represent the first temporally-consistent maps of the human footprint, as well as much more up-to-date information on cumulative pressures than is currently available. Moreover, the 2009 Human

Footprint map is the first cumulative pressure map to have undergone an accuracy assessment.

The individual pressure maps were developed to be globally consistent, using a scoring approach originally developed by Sanderson and colleagues (Sanderson et al. 2002). However, in some regions and for some species groups, alternate scores may be better suited for reflecting the pressures exerted by humans on nature. We therefore provide the individual pressure layers that compose the Human Footprint maps, thereby allowing data developers to create alternate scoring schemes that better suit their purposes, as well facilitating the addition of new or alternate data sources.

Moreover, our work is subject to three primary limitations. First, like all attempts to map cumulative pressures we did not fully account for all human pressures. Some of the missing and static pressures, such as invasive species and pollution, may be closely associated with pressures we did consider, and therefore their inclusion may not affect our overall results. Second, a lack of available data resulted in three of our pressures being static through time, which would cause an underestimation of Human Footprint expansion if these pressures expanded at a higher than average rate. Third, the Human Footprint measures the pressure humans place on nature, not the realized 'state' or 'impacts' on natural systems or their biodiversity. Significant scope exists to determine how natural systems respond to cumulating human pressures, and if non-linearity or thresholds exist where pressures lead to accelerated impacts.

While we welcome the opportunity to contribute intellectually and as co-authors to research projects that incorporate our datasets into their work, we make the data freely available without restriction for non-commercial use and redistribution. The data may be altered from their original form, and redistributed if done so free of charge and with a full description of any alterations to the original data. We do however ask that term 'Human Footprint map' be used only when referring to the unaltered data in the Human Footprint 7-zip file, and not to alternative versions of the data created by data users, and that the data be cited following the template at the end of this manuscript.

APPENDIX 2 Catastrophic declines in wilderness areas undermine global environment targets

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Contribution: JRA contributed to idea development, created the maps, analysed the data and helped write and edit the manuscript.

Summary

Humans have altered terrestrial ecosystems for millennia (Steffen et al. 2015a), yet wilderness areas still remain as vital refugia where natural ecological and evolutionary processes operate with minimal human disturbance (Lesslie et al. 1988, Watson et al. 2009, Kormos et al. 2016), underpinning key regional and planetary-scale functions (Gibson et al. 2011, Haddad et al. 2015). Despite the myriad values of wilderness areas – as critical strongholds for endangered biodiversity (Ripple et al. 2014), for carbon storage and sequestration (Houghton et al. 2015), for buffering and regulating local climates (Martin and Watson 2016), and for supporting many of the world's most politically and economically marginalized communities (Mackey and Claudie 2015) – they are almost entirely ignored in multilateral environmental agreements. This is because they are assumed to be relatively free from threatening processes and therefore are not a priority for conservation efforts (Myers et al. 2000, Mittermeier et al. 2003). Here we challenge this assertion using new comparable maps of global wilderness following methods established in the original ''last of the wild'' analysis (Sanderson et al. 2002) to examine the change in extent since the early 1990s. We demonstrate alarming losses comprising one-tenth (3.3 million km^2) of global wilderness areas over the last two decades, particularly in the Amazon (30%) and central Africa (14%). We assess increases in the protection of wilderness over the same time frame and show that these efforts are failing to keep pace with the rate of wilderness loss, which is nearly double the rate of protection. Our findings underscore an immediate need for international policies to recognize the vital values of wilderness and the unprecedented threats they face and to underscore urgent large-scale, multifaceted actions needed to maintain them.

Results and Discussion

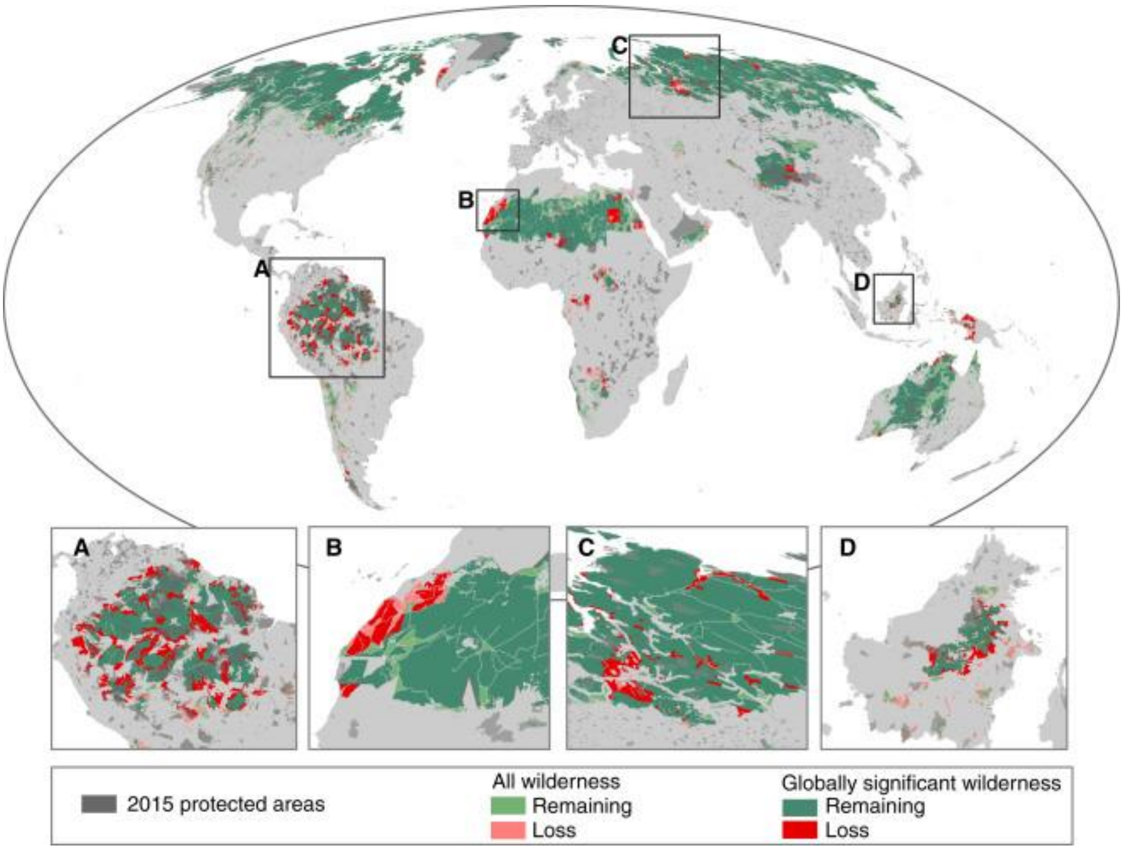
Contemporary Wilderness Loss

We mapped decline of wilderness areas, defining ‘‘wilderness’’ as biologically and ecologically largely intact landscapes that are mostly free of human disturbance (Lesslie et al. 1988, Mittermeier et al. 2003, Watson et al. 2009, Kormos et al. 2016). These areas do not exclude people, as many are in fact critical to certain communities, including indigenous peoples (Gorenflo et al. 2012, Schwartzman et al. 2013). Rather, they have lower levels of impacts from the kinds of human uses that result in significant biophysical disturbance to natural habitats, such as large-scale land conversion, industrial activity, or infrastructure development. We measured temporal change in wilderness extent by producing a global map of wilderness and assessing it against a spatially comparable map for the early 1990s (Figures A2.1). Both maps were devised using the same methodological framework as the original ‘‘last of the wild’’ map published in 2002 (Sanderson et al. 2002), but taking advantage of recently available datasets of in situ anthropogenic pressures. Following established practice, we exclude Antarctic and other ‘‘rock and ice’’ and ‘‘lake’’ ecoregions (Juffe-Bignoli et al. 2014, Venter et al. 2014b).

We discovered that a total of 30.1 million km^2 (or 23.2% of terrestrial areas) of the world’s land area now remains as wilderness, with the majority located in North America, North Asia, North Africa, and the Australian continent (Figures A2.1). An estimated 3.3 million km^2 has been lost since the early 1990s (approximately a 9.6% loss in two decades; Figure A2.2), with the most loss occurring in South America (experiencing 29.6% loss) and Africa (experiencing 14% loss).

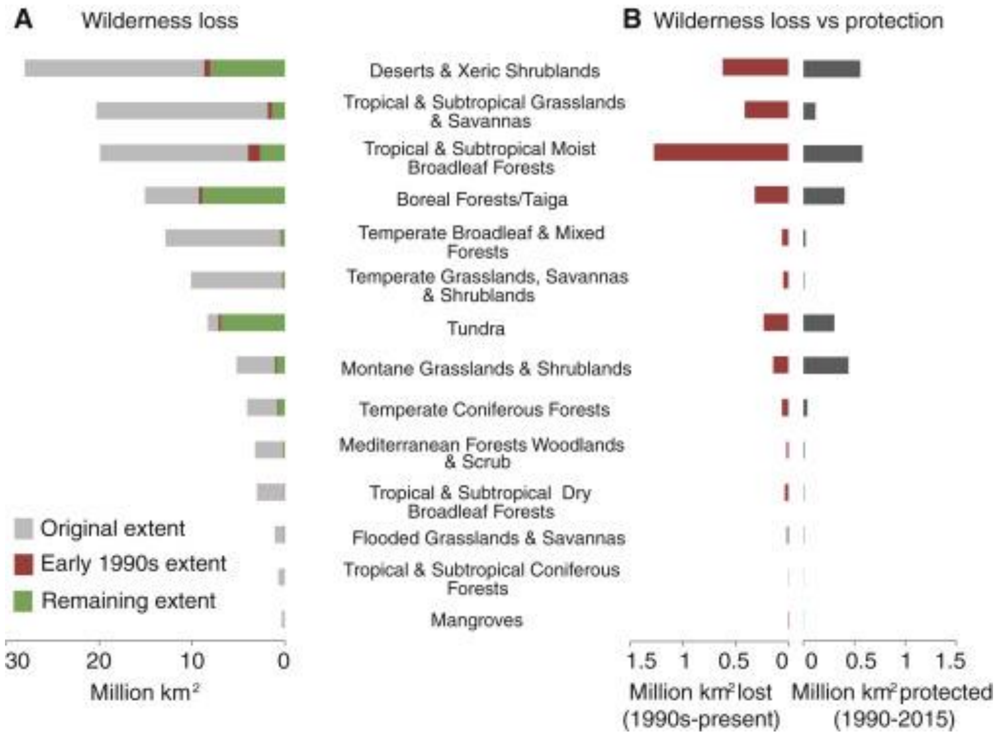
Encouragingly, the majority of wilderness (82.3%, or 25.2 million km^2) is still composed of large contiguous areas of at least 10,000 km^2. Although this is an arbitrary threshold, wilderness areas of this size are often considered as globally significant wilderness blocks (Mittermeier et al. 2003, Kormos et al. 2016). This is also the size threshold for identifying sites hosting intact ecological communities, adopted in the International Union for Conservation of Nature and Natural Resources (IUCN) standard for Key Biodiversity Areas (IUCN 2016). Yet there was substantial erosion of these large wilderness areas over the past two decades, with losses amounting to 2.7 million km^2 (Figure A2.1). A total of 37 of the 350 wilderness blocks that were present in the early 1990s have fallen below the area threshold used here for categorization as globally significant, and 74% of all blocks experienced erosion in areal extent. A total of 27 ecoregions (environmentally and ecologically distinct geographic units at the global scale (Olson et al. 2001)) have lost all of their remaining globally significant wilderness areas since the early 1990s, including those areas in the North-western Congolian Lowland Forests and the Northern New Guinea Lowland Rain and Freshwater Swamp Forests ecoregions. South America suffered particularly high losses in the Amazon basin, with the largest wilderness block being reduced from 1.8 million km2 to 1.3 million km^2 (a loss of over 30% in extent; Figures A2.1), and wilderness areas in the Ucayali Moist Forests and Iquitos Varzea ecoregions dropping below the globally significant threshold. This trajectory of wilderness loss in the Amazon is particularly concerning, given that overall deforestation rates reportedly dropped significantly across the Amazon Basin between 2005–2013 (Nepstad et al. 2014).

Figure A2.1 Change in the distribution of wilderness and globally significant wilderness areas since the early 1990s. Globally significant wilderness areas are defined as wilderness areas > 10,000 km^2. The insets are focused on the Amazon (A), the western Sahara (B), the West Siberian Taiga (C), and Borneo (D).



These recent losses have contributed further to existing biases in the geographical distribution of globally significant wilderness. Of Earth's 14 terrestrial biomes, three located mostly in the tropics (Tropical and Subtropical Coniferous Forests, Mangroves, and Tropical and Subtropical Dry Broadleaf Forests) now have no globally significant wilderness area remaining, with the last areas disappearing from two of these biomes over the last two decades. A further five biomes now have less than 10% wilderness remaining (Figure A2.2).

Figure A2.2. Historic and current extent of all wilderness area and the degree to which it is protected. (A) Historic (gray) and current (green) extent of all wilderness area lost since the early 1990s (red) across the world's terrestrial biomes. (B) The wilderness area lost (red) and protected (gray) during 1990-2015.



Disparity Between Wilderness Protection and Loss

Protected areas spearhead global efforts to conserve nature, and when properly managed they are particularly effective for combating the effects of habitat loss and degradation (CBD 2008). Since its inception, and through work plans such as the Aichi Targets of The Strategic Plan for Biodiversity 2011–2020 (CBD 2011), the Convention on Biological Diversity (CBD) has promoted protected areas as a vital conservation mechanism. Consequently, there has been a pronounced expansion of the global protected area estate over the past two decades, with its extent being an almost doubled since the Rio Earth Summit in 1992 (Juffe-Bignoli et al. 2014). However, despite this growth, the increase in protection of wilderness has lagged significantly behind losses over the past two decades: 2.5 million km^2 of wilderness areas (including 2.1 million km^2 considered globally significant) was newly protected, whereas 3.3 million km^2 (including 2.7 million km^2 considered globally significant) was lost. In some biomes, there has been a stark contrast between the area lost and the amount protected (Figure A2.2). For example, the Mediterranean Forests, Woodlands, and Scrub biome lost 37% of its globally significant wilderness extent since the early 1990s, yet there was no reciprocal protection of the remaining wilderness areas. Similarly, 23% of the globally significant wilderness was lost from the Tropical and Subtropical Grasslands, Savannas, and Shrublands, with only 8.5% protected in the last two decades.

Consequences of Continued Wilderness Loss

The current levels of non-protection and consequent loss of wilderness areas across the planet have important ramifications for achieving global climate mitigation goals (Houghton et al. 2015). For example, the total stock of terrestrial ecosystem carbon (~1,950 petagrams of Carbon [Pg C]) is greater than that of oil (~173 Pg C), gas (~383 Pg C), coal (~446 Pg C), or the atmosphere (~598 Pg C) (Ciais and Sabine 2013), and a significant proportion of this carbon is found in the globally significant wilderness areas of the tropics and boreal region (Pan et al. 2011, Houghton et al. 2015). It is estimated that 32% of the total global stock of forest biomass carbon is stored in the boreal forest biome (Pan et al. 2011) and that the Amazon region stores nearly 38% (86.1 Pg C) of the carbon (228.7 Pg C) found above ground in the woody vegetation of tropical America, Africa, and Asia (Walker et al. 2014). Thus, avoiding emissions by protecting the globally significant wilderness areas of the boreal and Amazon in particular will make a significant contribution to stabilizing atmospheric concentrations of CO^2. Protection of intact forest ecosystems from industrial land uses is particularly important, given that they store more carbon than degraded forests and are more resilient to external perturbations, including climate variability, fire, and illegal logging, poaching, and mining (Thompson et al. 2009, Houghton et al. 2015).

Although both the boreal and Amazon have suffered significant forest loss and degradation, these landscapes still support globally significant wilderness areas and are increasingly threatened by industrial forestry, oil and gas exploration, anthropogenic fire, and rapid climate change. If allowed to continue unchecked, these impacts will result in depletion of ecosystem carbon stocks and significant CO^2 emissions, converting the biome into a large carbon source (Bradshaw et al. 2009). For example, on Borneo and Sumatra in 1997, human-induced fires burned into recently converted wilderness areas harbouring large peat carbon stores, causing the release of over 1 Pg C (Page et al. 2002), which is equivalent to about 10% of all annual anthropogenic CO^2 emissions (Le Quéré et al. 2015).

In terms of biodiversity values, an analysis of the IUCN Red List for terrestrial mammals – one of the taxonomic groups that has been most completely assessed – shows that Earth's remaining wilderness areas also sustain the last strongholds of many imperilled species. The geographic ranges of one-third of all terrestrial mammal species overlap with globally significant wilderness areas, including extensive parts of the distribution of 12% (143) of all threatened mammal species. Thus, ongoing and rapid loss of wilderness increases the risk of extinction for species that are already highly threatened. It is also well established that wilderness areas are critical for wide-ranging and migratory species reliant on intact ecosystems (and their associated ecological processes) and represent residual habitats for disturbance-sensitive species and for those that have a conflictual coexistence with humans, such as many of the world's large carnivores (Crooks et al. 2011).

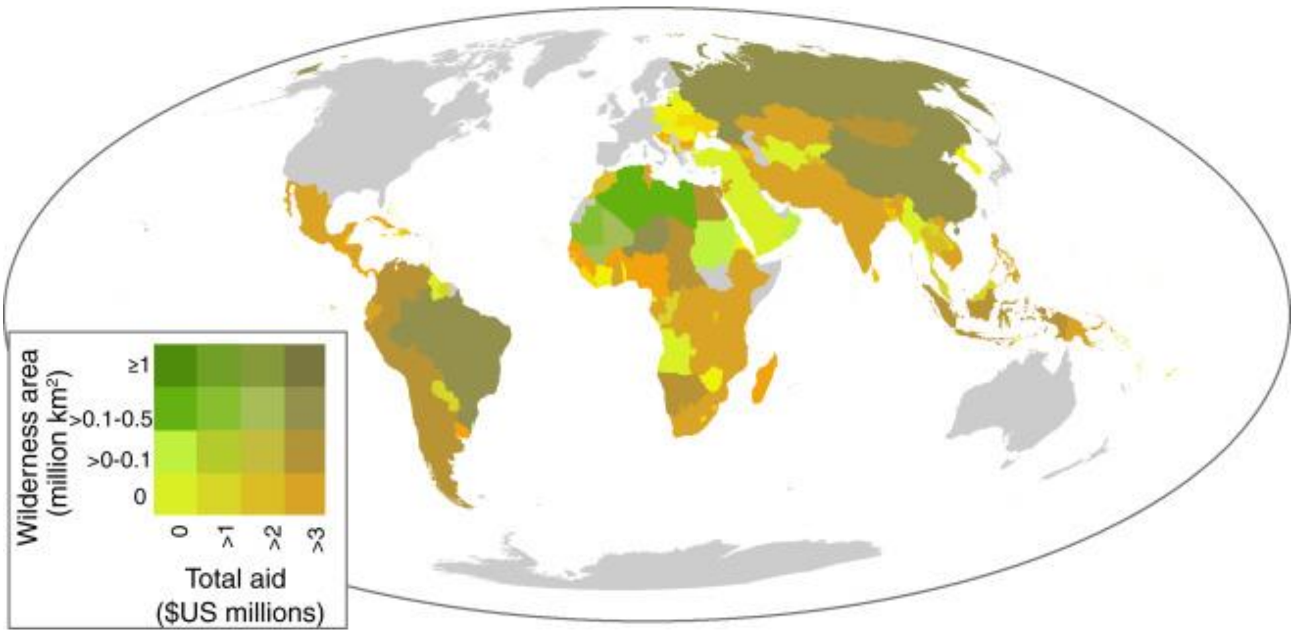
Wilderness areas also provide benefits derived from their large-scale and self-organization (Sanderson et al. 2002), and in many instances they are likely to operate as entire systems, where losses in one area inevitably affect long-term environmental outcomes in another (Laurance 2005, Peres 2005, Cochrane and Laurance 2008). For example, in the Amazon, it is thought that at least 60% of the forest cover is required to maintain the hydrological cycle (Sampaio et al. 2007), and so conservation action at the scale of the whole ecosystem is required to ensure that this large wilderness area is maintained. In Australian rangeland and desert ecosystems, the ecological influence of large spatial-scale surface-groundwater hydrological dynamics is pervasive, and losses in one area can degrade habitat quality elsewhere, with significant, long-term implications for biodiversity (Soule et al. 2004b). In the Anthropocene era, where the human footprint is now altering many of Earth systems processes (Venter et al. 2016c), wilderness areas serve as natural observatories where we can study the ecological and evolutionary impacts of global change. They also serve as natural controls for comparison with areas where intensifying land use and land cover changes are occurring. As intact, large-scale ecosystems become rarer, their value is increasingly appreciated. For instance, we are already seeing growing efforts to ''rewild'' some human-dominated ecosystems in Europe and North America (Navarro and Pereira 2012); remaining wilderness areas provide the reference points and biological feedstock for these initiatives. Without concerted preservation of existing wilderness areas, there will be a diminished capacity for large-scale ecological restoration.

Implications for Multilateral Environmental Agreements

The recent severe loss of wilderness is impacting options for achieving strategic goals outlined in key multilateral environmental agreements, including the CBD's 2020 Aichi Targets and the United Nations Framework on Climate Change (UNFCCC) Paris Agreement (CBD 2011, United Nations 2015b). There are a number of reasons why globally significant wilderness areas are ignored in policy deliberations. International definitions of forests have not differentiated between types of forests and in some cases actually treat primary forests, degraded forests, and plantations as equivalent (Mackey et al. 2015). International polices do not acknowledge the special qualities and benefits that flow from ecosystem processes operating at large scales. For example, there is no formal text within the UNFCCC, United Nations World Heritage Convention (WHC), or CBD that prioritizes or even recognizes the benefits derived from large intact landscapes for nature and people. An emphasis on degraded, fragmented, and altered ecosystems has ramifications for national environmental strategies. The tendency is to focus national biodiversity conservation plans on remnant habitats and endangered populations (Watson et al. 2009, Ceauşu et al. 2015), with few nations clearly articulating conservation goals for wilderness area.

The lack of recognition of wilderness in global accords and national policy also has implications for international funding programs such as the Global Environment Facility, Green Climate Fund, and Critical Ecosystems Partnership Fund, which are distributing billions of dollars in support for programs to help achieve the goals of multilateral environmental agreements. Within the CBD funding mechanisms, for example, 80% of funds have been allocated to nations with <20% of all wilderness area (Figure A2.3). The neglect of wilderness is arguably even more acute in funding under the UNFCCC and Paris-

Figure A2.3. Amount of conservation aid and extent of wilderness now remaining per country. Amount of conservation aid is shown in million km^2. Gray areas indicate countries that received no aid.



Agreement finance discussions. Although there is strong financing for forest conservation under the UNFCCC REDD+ mechanism to reduce emissions from deforestation and degradation, the rules stipulate that this financing must target areas with high baseline levels of deforestation (Laurance et al. 2014). Such efforts, though valuable for other purposes, serve to direct funds away from forested wilderness areas that are presumed safe from deforestation and degradation. As our results demonstrate, however, wilderness is under immense land use pressures, and there is an urgent need for greater conservation effort in these areas to help maintain their ecological intactness and integrity of function.

What would it take to halt the rapid loss of wilderness and of globally significant areas in particular? Achieving meaningful changes in policy at the global level is more likely if there is first a critical mass of support at the national level. Ideally, this should be evidenced through national strategies and plans that recognize the values of wilderness areas and specify policies for their protection. In any case, by creating clear text within operational guidelines, work plans, and ongoing negotiations of key multilateral environmental agreements, international conservation investments can then be mobilized and focused in a manner that can fund activities to help protect wilderness areas. These activities will vary based on the specific context of different nations and regions, but there is a clear need to focus on halting current threatening activities that have been leading to the recent erosion of wilderness areas, including limiting road expansion (Laurance et al. 2014); preventing industrial mining, forestry, and other large-scale agricultural operations (Edwards et al. 2014, Laurance et al. 2014); and enforcing existing legal frameworks considering that half of all tropical forest clearing between 2000 and 2012 was illegal (Fitzherbert et al. 2008, Lawson et al. 2014). A key goal could be to proactively fund conservation interventions in wilderness areas where degrading activities are currently absent but are projected to occur in the near future.

Conservation actions should include (1) creating large and, where necessary, multi-jurisdictional protected areas; (2) establishing mega-conservation corridors between protected areas; and (3) enabling indigenous communities to establish community conservation reserves (Schwartzman et al. 2013). Funding could also be used to establish payments for ecosystem service programs that recognize the direct and indirect economic benefits that wilderness areas provide, such as being a secure source of fresh water, reducing disaster risks, and storing large carbon stocks (Martin and Watson 2016). There are some encouraging examples where these types of activities are being undertaken. For example, in Brazil, the Amazon Region Protected Areas (ARPA) program supports the creation and management of protected areas and sustainable natural resource management reserves (WWF 2016). The overarching aim of these protected areas and reserves is to maintain forest carbon stocks, protect large-scale ecological processes, and establish sustainable use by local peoples. This program is now extending beyond Brazil to Peru and Colombia. The Canadian Boreal Forest Conservation Framework is a similar example, with an overall aim of conserving the long-term integrity of the boreal forest biome by protecting at least 50% of the Boreal in a network of large interconnected protected areas and supporting sustainable communities via ecosystem-based resource management and stewardship practices across the remaining landscape (Boreal Leadership Council 2003).

These positive examples are too few, and we argue that immediate action to protect the world’s remaining wilderness areas on a large scale is now necessary, including in global policy platforms. All wilderness areas, regardless of their size threshold, warrant immediate scrutiny for conservation action, especially in regions with low levels of remaining wilderness areas. The continued loss of wilderness areas is a globally significant problem with largely irreversible outcomes for both humans and nature: if these trends continue, there could be no globally significant wilderness areas left in less than a century. Proactively protecting the world’s last wilderness areas is a cost-effective conservation investment and our best prospect for ensuring that intact ecosystems and large-scale ecological and evolutionary processes persist for the benefit of future generations.

APPENDIX 3 Supplementary material

Chapter 3 Supplementary Tables and Figures for "Hotspots of human impacts on threatened terrestrial vertebrates"

Fig. S3.1. Impact hotspots of individual human pressures on all threatened terrestrial vertebrates (n=5457), mammals (n = 1277), birds (n = 2120), and amphibians (n=2060). Scale represents the number of species that are impacted by the threat in a grid cell. Hotspots of impact are shown in dark red. Maps use a 30x30 km grid and a Mollweide equal area projection.

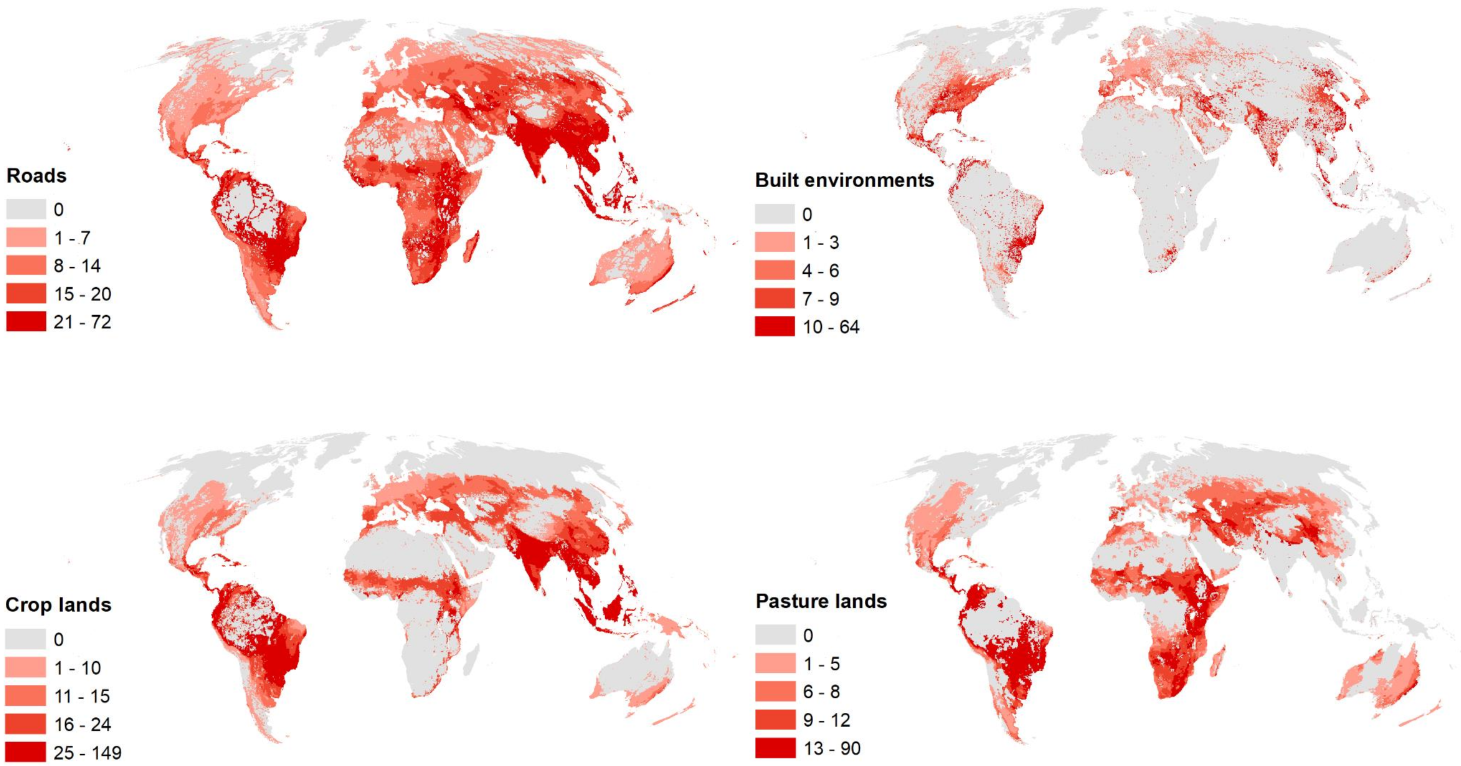


Fig. S3.2. Impact hotspots of individual human pressures on all threatened terrestrial vertebrates (n=5457), mammals (n = 1277), birds (n = 2120), and amphibians (n=2060). Scale indicates the number of species that are impacted by the threat in a grid cell. Hotspots of impact are shown in dark red. Maps use a 30x30 km grid and a Mollweide equal area projection.

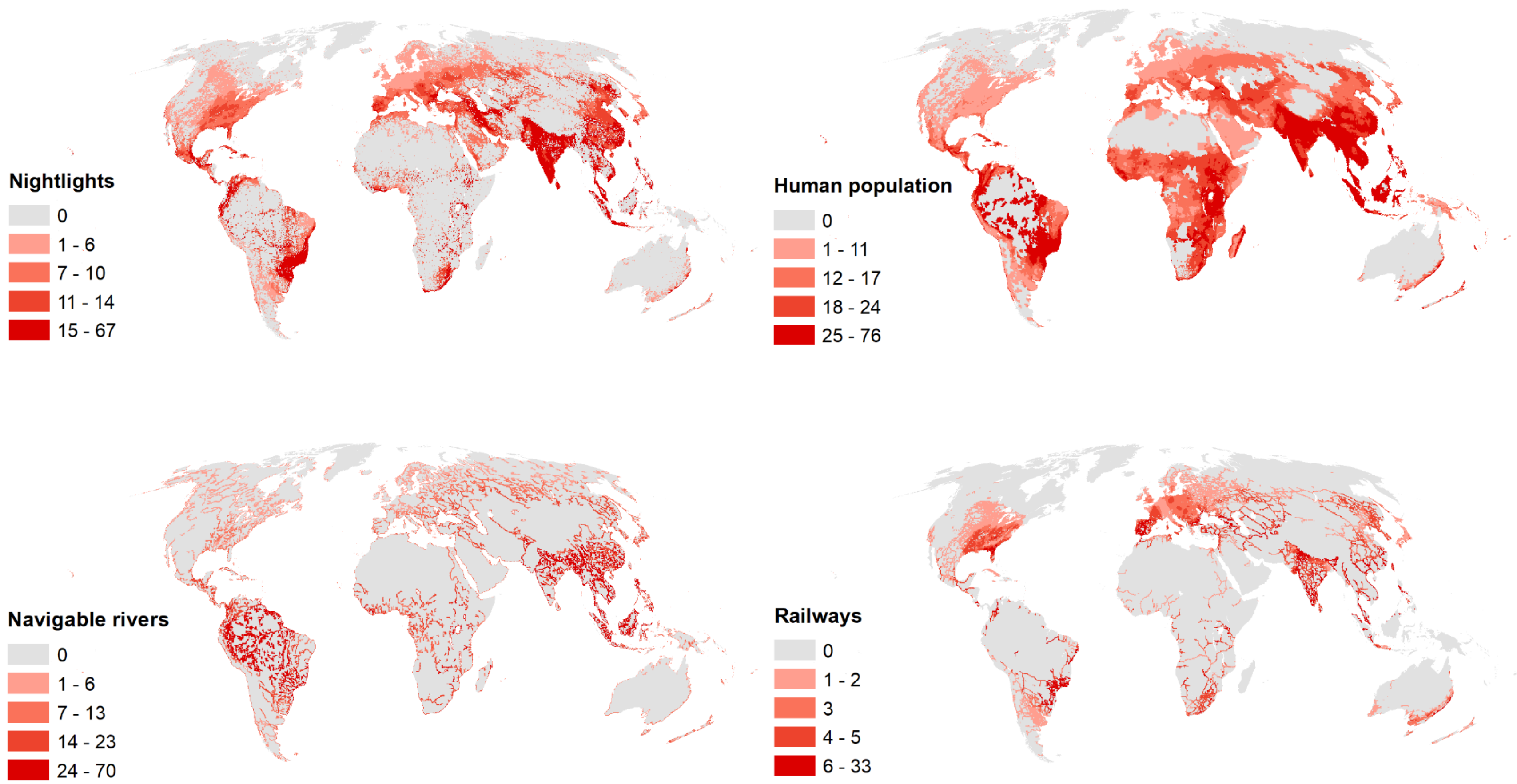


Fig. S3.3. Cumulative human impacts on all threatened terrestrial vertebrates (n=5457), mammals (n = 1277), birds (n = 2120), and amphibians (n=2060). Scale indicates the number of species in a grid cell impacted by at least one threat. Areas of high human impact (hotspots) are shown in Red. Maps use a 30x30 km grid and a Mollweide equal area projection.

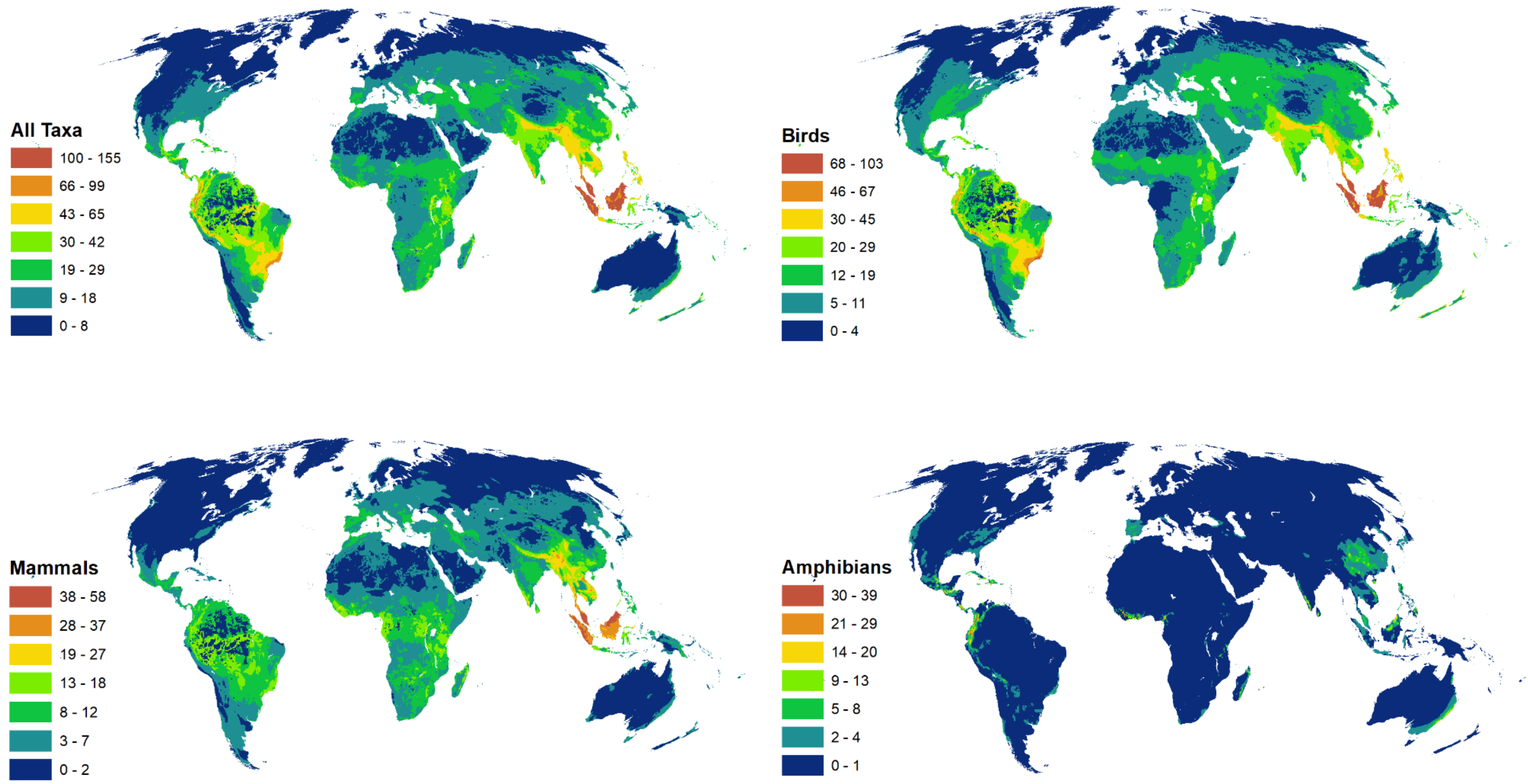


Fig. S3.4. Threatened species richness for all taxa (n=5457), mammals (n = 1277), birds (n = 2120), and amphibians (n=2060. Areas of high human richness are shown in Red. Maps use a 30x30 km grid and a Mollweide equal area projection.

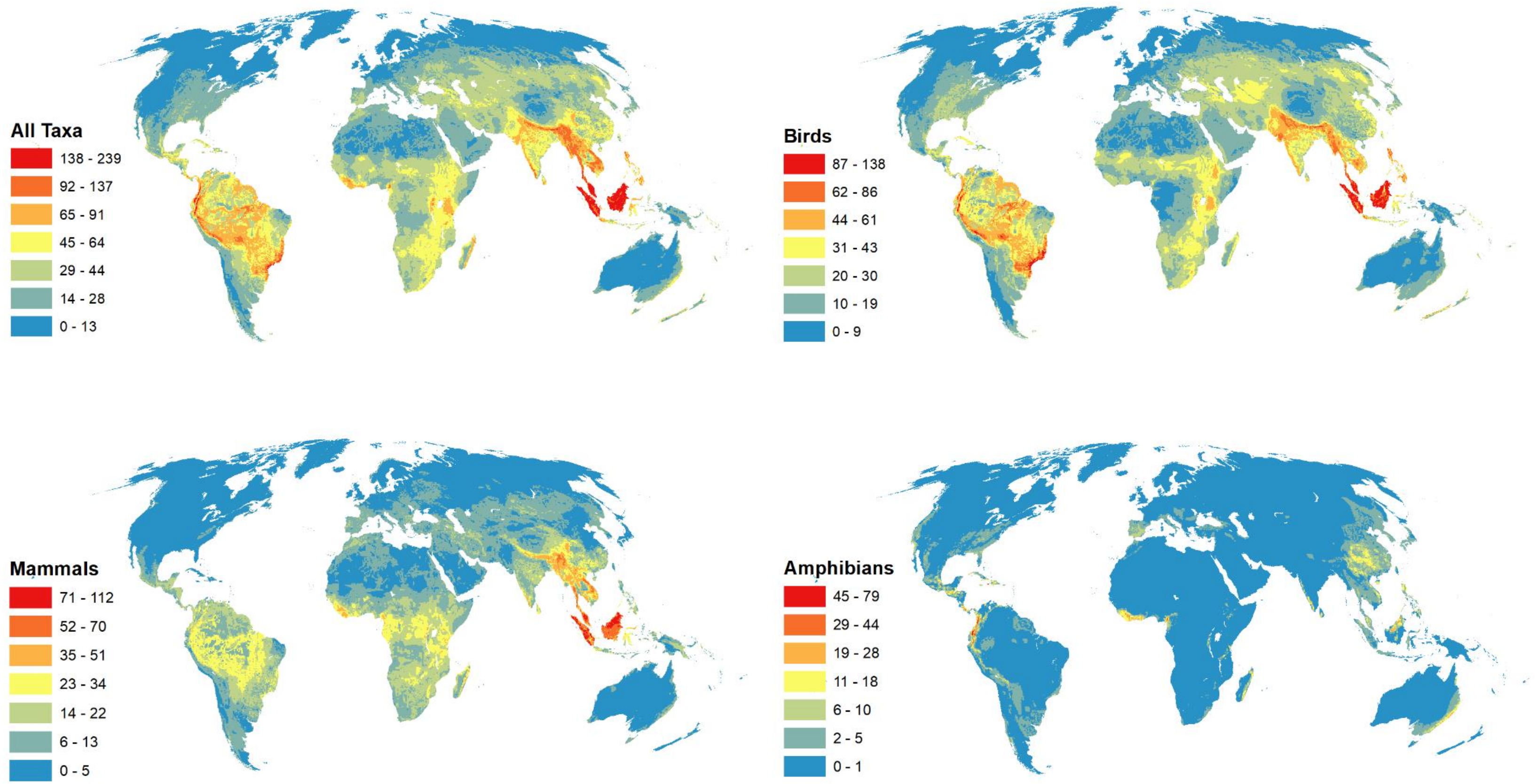


Fig. S3.5. Coolspots of refugia for all threatened terrestrial vertebrates (n=5457), mammals (n = 1277), birds (n = 2120), and amphibians (n=2060). Scale indicates the number of species that are not impacted by any threats in a grid cell. Coolspots of refugia are shown in Blue. Maps use a 30x30 km grid and a Mollweide equal area projection.

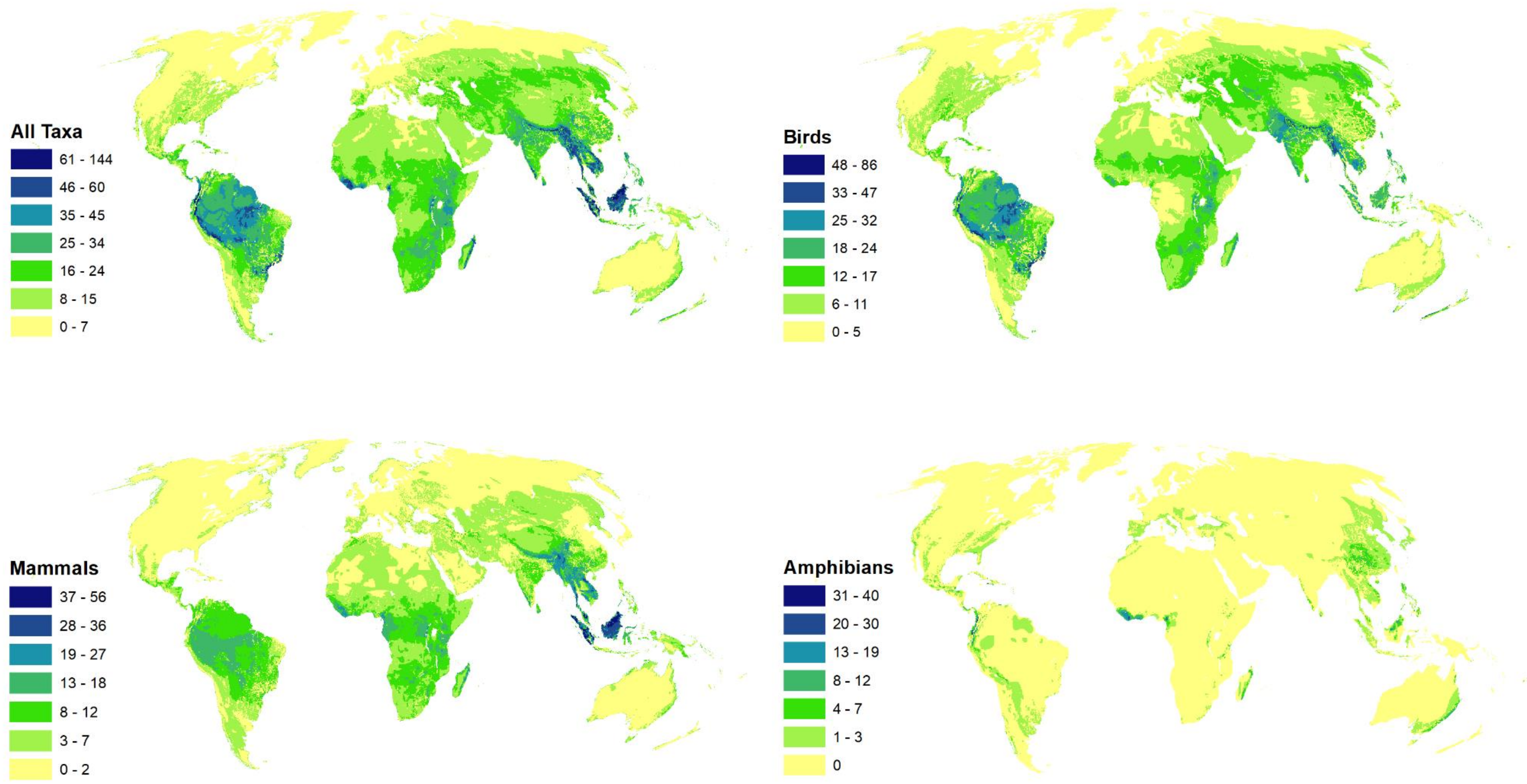


Table S3.1. The eight mapped human pressures, the number of sensitive species they are impacting, the area in which these impacts are occurring, and the proportion of Earth's terrestrial area where these impacts are occurring.

Human Pressures (threats)	Number of sensitive species Impacted	Area where sensitive species are impacted (km2)	Proportion of Earth's terrestrial area where impacts are occurring (%)
Roads	2832	103,873,500	72
Crop lands	3834	65,234,800	45
Pasture lands	1642	58,348,800	40
Built Environments	1565	27,027,900	18
Nightlights	2049	50,246,100	35
Navigable Waterways	1531	35,118,000	24
Railways	405	27,903,600	19
Population Density	2856	82,519,200	57

Table S3.2. The top ten countries with the most impacted and unimpacted species on average.

Highly impacted	**Mean number of impacted species per grid cell**	**Mean number of unimpacted species per grid cell**
Malaysia	125	48
Brunei	124	49
Singapore	112	5
Indonesia	69	31
Myanmar	52	44
Ecuador	50	34
Cambodia	47	43
Thailand	47	29
Laos	46	39
Bhutan	45	38
Highly unimpacted		
Brunei	124	49
Liberia	26	48
Malaysia	125	48
Myanmar	52	44
Cambodia	47	43
Laos	46	39
Bhutan	45	38
Suriname	19	36
Nepal	43	36
Guyana	30	34

Table S3.3. The average number of species impacted and unimpacted by threats per grid cell, and the proportion of species impacted by threats per grid cell, in each of Earth's Biomes.

Biome Name	Average number of species impacted per grid cell	Average number of species not impacted per grid cell	Average proportion of species impacted per grid cell
Mangrove	35.0	20.7	61.3
Temperate broadleaf and mixed forests	14.7	10.2	60.7
Mediterranean forests woodlands and scrub	13.9	8.9	60.66
Tropical and subtropical dry broadleaf forests	29.8	21.3	60.44
Tropical and subtropical coniferous forests	21.7	15.2	59.88
Temperate grasslands savannas and shrublands	13.1	10.5	56.39
Flooded grasslands and savannas	25.0	21.0	55.7
Tropical and subtropical moist broadleaf forests	34.8	28.8	51.46
Tropical and subtropical grasslands savannas	19.6	17.8	51.2
Temperate coniferous forests	11.6	9.7	49.146
Montane grasslands and savannas	16.5	15.8	47.89
Deserts and xeric shrublands	10.1	10.8	42.99
Boreal forests taiga	3.5	5.7	29.03
Tundra	1.2	3.7	14.62

Extended Data Table 3.4. Weights assigned to individual pressures in the Human Footprint, and threshold scheme used to convert pressures into binary scores (present or absent) for impact analyses.

Pressure	Score	Details	Threshold for conversion to binary
Built environments	0,10	All built areas given score of 10	pressure present or absent
Population density	0-10 Continuous	Score = 3.333xlog(population density+1)	Pressure considered present for scores ≥ 1.
Night-time lights	0-10 Continuous	Equal quintile bins	Pressure considered present for scores ≥ 1.
Croplands	0,7	All cropland given score of 7	pressure present or absent
Pasture	0,4	All pasture given score of 4	pressure present or absent
Roads	0,8 Direct impacts 0-4 indrect impacts	500m either side of road given a direct pressure score of 8. Starting 500m out from road, pressure score of 4 exponentially decaying out to 15km	Pressure considered present up to 3km either side of the road (equivalent human footprint score = 1)
Railways	0,8	500m either side of railway given a direct pressure score of 8	pressure present or absent
Navigable waterways	0-4	pressure score of 4 exponentially decaying out to 15km	Pressure considered present up to 1.5 km either side of the waterway

	(equivalent human footprint score = 3.5)

Chapter 4 Supplementary Tables for "Recent increases in human pressure and forest loss threaten many Natural World Heritage Sites

Table S4.1. Change in the Human Footprint between 1993 and 2009 in Natural World Heritage Sites (NWHS) and 10km Buffer zones. Only NWHS inscribed prior to 1993 are included.

		Human Footprint 1993		Change in Human Footprint		Human Footprint 2009	
WDPAID	English Name	NWHS	Buffer	NWHS	Buffer	NWHS	Buffer
67727	Air and Tenere Natural Reserves	0.2	0.6	0.1	0.3	0.3	1.0
20388	Banc d'Arguin National Park	2.3	1.3	0.0	0.0	2.3	1.3
2008	Bialowieza Forest	12.6	9.7	-4.1	1.2	8.5	10.8
26689	Canadian Rocky Mountain Parks	1.3	1.5	-0.1	-0.1	1.3	1.4
10905	Chitwan National Park	11.5	13.9	3.0	3.5	14.5	17.5
20171	Cliff of Bandiagara (Land of the Dogons)	8.4	7.9	0.6	0.5	9.1	8.5
9545	Comoe National Park	5.0	6.1	0.3	0.3	5.2	6.4

67728	Danube Delta	6.8	13.9	-2.3	0.2	4.5	13.9
2554	Darien National Park	2.4	3.2	1.3	1.7	3.8	5.0
2004	Dinosaur Provincial Park	2.4	4.6	0.1	0.2	2.5	4.8
17758	Dja Faunal Reserve	2.4	3.9	0.9	0.6	3.4	4.5
2578	Djoudj National Bird Sanctuary	8.2	8.6	0.6	0.5	8.8	9.3
4326	Durmitor National Park	10.1	10.6	-0.2	-0.6	9.8	9.9
2012	Everglades National Park	4.3	5.8	-0.9	-0.5	3.5	5.2
67730	Fraser Island	8.8	7.8	0.6	0.2	9.4	8.1
12206	Garajonay National Park	10.7	12.3	-0.3	0.9	10.6	13.4
4327	Garamba National Park	3.7	4.4	0.0	0.8	3.7	5.3
12202	Gondwana Rainforests of Australia	2.6	4.7	0.3	0.5	2.9	5.2
478637	Goreme National Park and the Rock Sites of Cappadocia	22.0	13.2	-3.3	0.0	18.8	12.9
2011	Grand Canyon National Park	5.2	4.2	0.0	0.1	5.1	4.4
9632	Great Smoky Mountains National Park	5.3	10.6	0.1	0.3	5.4	10.9
17759	Gros Morne National Park	2.5	2.3	-0.1	-0.1	2.4	2.1
12896	Henderson Island	1.0	na	0.0	na	1.0	na
478640	Hierapolis-Pamukkale	23.5	14.6	-6.5	-0.2	17.0	14.3
9623	Historic Sanctuary of Machu Picchu	9.1	8.9	0.7	0.1	9.7	9.0
67733	Huanglong Scenic and Historic Interest Area	5.3	5.7	0.7	1.0	6.0	6.8
10747	Huascaran National Park	5.7	6.9	0.0	0.4	5.6	7.3

4322	Ichkeul National Park	12.3	19.1	-0.7	1.5	11.3	20.7
12203	Iguacu National Park	7.0	7.8	0.2	4.5	7.2	12.5
10901	Iguazu National Park	8.7	10.0	0.5	1.5	9.2	11.5
67732	Jiuzhaigou Valley Scenic and Historic Interest Area	5.0	5.4	0.9	1.3	5.9	6.7
4328	Kahuzi-Biega National Park	6.0	7.1	-0.2	0.3	5.8	7.4
2572	Kakadu National Park	0.8	0.5	0.0	0.0	0.9	0.5
10744	Kaziranga National Park	10.6	12.3	2.0	1.9	12.6	14.2
10746	Keoladeo National Park	19.9	21.9	1.6	1.9	21.4	23.8
17761	Kilimanjaro National Park	7.7	11.3	0.3	-2.0	8.0	9.2
2018	Kluane / Wrangell-St Elias / Glacier Bay / Tatshenshini-Alsek	0.4	1.4	0.0	-0.1	0.3	1.3
67725	Komodo National Park	6.2	na	4.3	na	10.6	na
10904	Lake Malawi National Park	13.6	11.1	-0.5	0.1	13.1	11.4
5001	Lord Howe Island Group	4.0	na	0.0	na	4.0	na
2570	Los Glaciares National Park	0.7	1.3	0.1	0.1	0.7	1.4
2577	Mammoth Cave National Park	5.6	10.9	0.1	-0.2	5.8	11.1
10907	Mana Pools National Park, Sapi and Chewore Safari Areas	9.0	8.9	-2.9	-2.2	6.2	6.7
10745	Manas Wildlife Sanctuary	11.8	12.0	5.3	2.2	17.0	14.2
16792	Manovo-Gounda St Floris National Park	0.9	1.3	0.9	0.8	1.8	2.1
17760	Manu National Park	0.5	2.0	0.2	1.0	0.8	3.0

20399	Mosi-oa-Tunya / Victoria Falls	15.3	13.2	-0.5	-1.1	14.9	12.2
18863	Mount Athos	11.2	15.2	-0.9	0.7	10.2	16.0
26654	Mount Huangshan	9.7	10.6	0.4	-0.5	10.0	10.1
2574	Mount Nimba Strict Nature Reserve	7.9	8.8	0.8	0.1	8.7	8.8
17050	Mount Taishan	16.0	19.4	1.1	3.3	17.1	22.6
2005	Nahanni National Park	0.7	0.5	-0.5	-0.3	0.1	0.1
16793	Nanda Devi and Valley of Flowers National Parks	2.3	4.3	0.3	1.2	2.7	5.5
2015	Natural and Cultural Heritage of the Ohrid region	14.7	11.7	-1.2	-1.0	13.5	10.6
2010	Ngorongoro Conservation Area	5.0	6.0	0.4	-0.1	5.4	6.0
2580	Niokolo-Koba National Park	3.2	4.1	0.0	0.7	3.1	4.8
2579	Olympic National Park	1.3	4.0	0.2	0.1	1.5	4.1
9613	Pirin National Park	9.4	11.8	-1.2	-1.8	8.2	9.8
2016	Plitvice Lakes National Park	5.6	7.6	0.4	0.1	6.0	7.9
26651	R¡o Abiseo National Park	2.1	2.8	0.2	0.9	2.3	3.7
5002	R¡o Platano Biosphere Reserve	1.4	2.6	1.2	0.3	2.7	3.0
4325	Redwood National and State Parks	5.7	5.9	0.0	-0.1	5.7	5.6
2007	Sagarmatha National Park	6.3	3.8	0.2	-0.1	6.5	3.7
10906	Salonga National Park	2.3	3.3	0.5	0.6	2.8	3.9
9614	Sangay National Park	1.1	5.7	0.5	1.0	1.7	6.6
5005	Selous Game Reserve	5.5	6.2	-0.1	0.2	5.4	6.3

2575	Serengeti National Park	7.6	7.9	-0.8	-0.5	6.9	7.4
67724	Shark Bay, Western Australia	4.8	6.0	0.6	0.2	5.4	6.3
68915	Shirakami-Sanchi	7.1	9.8	0.8	0.8	8.0	10.6
20062	Sian Ka'an	6.8	5.8	-1.9	-0.8	4.9	5.0
2006	Simien National Park	5.7	8.2	2.9	2.2	8.6	10.1
16791	Sinharaja Forest Reserve	16.7	17.7	-7.0	-6.3	9.7	11.5
9612	Srebarna Nature Reserve	13.6	16.4	2.2	0.6	15.8	17.2
902368	St Kilda	4.9	na	3.5	na	8.4	na
14177	Sundarbans National Park	9.1	11.2	-0.1	-0.4	9.1	10.8
5003	Tai National Park	3.9	5.0	1.3	1.5	5.2	6.5
10903	Talamanca Range-La Amistad Reserves / La Amistad National Park	4.2	7.5	0.8	1.1	5.0	8.6
5000	Tasmanian Wilderness	1.2	3.1	0.2	0.4	1.4	3.5
4999	Tassili n'Ajjer	1.4	2.0	0.0	0.0	1.4	2.1
26652	Te Wahipounamu South West New Zealand	2.1	5.0	0.2	1.0	2.4	5.9
67729	Thungyai - Huai Kha Khaeng Wildlife Sanctuaries	4.2	6.4	-0.1	-1.0	4.2	5.4
197	Tikal National Park	2.8	2.4	1.0	0.4	3.8	2.9
26649	Tongariro National Park	3.3	4.2	0.0	1.4	3.3	5.6
26653	Tsingy de Bemaraha Strict Nature Reserve	2.2	4.3	0.8	0.6	3.0	4.9
67726	Ujung Kulon National Park	8.7	18.0	-1.0	-3.1	7.8	14.9

900010	Uluru-Kata Tjuta National Park	1.8	0.7	0.0	0.0	1.8	0.7
2017	Virunga National Park	11.1	11.3	-0.5	-0.9	10.5	10.3
17757	Wet Tropics of Queensland	3.9	5.6	0.4	0.3	4.2	5.9
68918	Whale Sanctuary of El Vizcaino	3.7	3.7	0.1	0.2	3.8	3.9
10902	Wood Buffalo National Park	0.4	1.1	0.1	0.0	0.5	1.2
67731	Wulingyuan Scenic and Historic Interest Area	7.6	10.5	1.4	2.8	8.9	13.3
68916	Yakushima	9.1	11.5	-0.3	0.3	8.9	11.8
2013	Yellowstone National Park	1.4	1.2	-0.1	0.0	1.3	1.2
10908	Yosemite National Park	2.7	2.3	0.1	0.2	2.8	2.5

Table S4.2. Total forest loss (km2), and forest loss as a percentage of forested area, between 2000 and 2012 in Natural World Heritage Sites (NWHS) and buffer zones. Only NWHS inscribed prior to 2000 are included.

		Total forest loss 2000 - 2012 (km2)		Percentage of forested area lost 2000 - 2012	
WDPAID	English name	NWHS	Buffer	NWHS	Buffer
67727	Air and Tenere Natural Reserves	0.0	0.0	0.0	0.0
198296	Area de Conservacion Guanacaste	4.3	52.0	0.7	4.6
198293	Atlantic Forest Southeast Reserves	18.6	225.3	0.4	2.4
61604	Australian Fossil Mammal Sites (Riversleigh / Naracoorte)	0.0	9.6	1.8	32.9
2008	Bialowieza Forest	23.6	24.9	2.2	3.2
61609	Bwindi Impenetrable National Park	0.8	21.4	0.3	3.6
26689	Canadian Rocky Mountain Parks	424.5	176.4	5.3	3.7
61612	Canaima National Park	105.7	86.0	0.5	1.3
93291	Carlsbad Caverns National Park	0.3	0.7	7.4	4.0
220296	Central Amazon Conservation Complex	51.9	48.8	0.1	0.3
220298	Central Suriname Nature Reserve	10.4	2.8	0.1	0.0
10905	Chitwan National Park	1.7	6.3	0.2	0.5

20171	Cliff of Bandiagara (Land of the Dogons)	0.0	0.0	0.0	0.0
9545	Comoe National Park	17.9	27.1	0.9	3.1
67728	Danube Delta	1.4	8.3	0.2	2.1
2554	Darien National Park	10.3	89.5	0.2	2.0
198297	Desembarco del Granma National Park	5.1	4.7	2.6	3.2
2004	Dinosaur Provincial Park	0.0	0.0	0.0	1.0
198292	Discovery Coast Atlantic Forest Reserves	19.2	192.5	1.7	11.4
17758	Dja Faunal Reserve	1.9	14.4	0.0	0.4
2578	Djoudj National Bird Sanctuary	0.0	0.0	0.2	0.0
61611	Donana National Park	2.1	1.0	7.3	0.8
4326	Durmitor National Park	1.9	6.8	1.5	1.0
168242	East Rennell	3.0	36.7	2.1	1.0
2012	Everglades National Park	36.2	2.9	3.1	0.7
67730	Fraser Island	11.5	6.1	1.1	5.3
12206	Garajonay National Park	0.1	0.4	0.4	1.0
4327	Garamba National Park	5.3	12.5	0.2	0.5
168241	Golden Mountains of Altai	98.0	36.7	2.6	1.0
12202	Gondwana Rainforests of Australia	2.5	73.3	0.1	0.8
478637	Goreme National Park and the Rock Sites of Cappadocia	0.0	0.0	0.0	0.9

2011	Grand Canyon National Park	38.2	5.1	9.8	1.1
9632	Great Smoky Mountains National Park	8.0	33.5	0.4	1.6
220294	Greater Blue Mountains Area	10.0	138.4	0.1	3.1
17759	Gros Morne National Park	1.8	26.8	0.2	2.8
220293	Gunung Mulu National Park	0.5	25.0	0.1	2.0
900889	Ha Long Bay	0.0	3.6	0.0	2.2
478640	Hierapolis-Pamukkale	0.0	1.0	0.0	4.4
9623	Historic Sanctuary of Machu Picchu	0.4	2.6	0.4	0.8
67733	Huanglong Scenic and Historic Interest Area	0.2	0.7	0.2	0.3
10747	Huascaran National Park	0.1	0.8	0.2	0.5
4322	Ichkeul National Park	0.0	3.9	0.0	6.7
12203	Iguacu National Park	0.1	50.3	0.0	5.5
10901	Iguazu National Park	0.2	110.4	0.0	12.8
220291	Ischigualasto / Talampaya Natural Parks	0.0	0.0	0.1	0.0
198302	iSimangaliso Wetland Park	19.6	161.9	4.6	17.9
67732	Jiuzhaigou Valley Scenic and Historic Interest Area	0.1	0.5	0.1	0.1
4328	Kahuzi-Biega National Park	57.0	182.4	0.9	3.5
2572	Kakadu National Park	0.8	0.7	0.0	0.1
10744	Kaziranga National Park	2.5	13.0	2.1	3.5
10746	Keoladeo National Park	0.4	0.0	16.6	5.1

17761	Kilimanjaro National Park	0.6	22.0	0.5	2.4
220292	Kinabalu Park	2.3	150.0	0.3	9.7
2018	Kluane / Wrangell-St Elias / Glacier Bay / Tatshenshini-Alsek	139.8	32.6	1.1	0.6
67725	Komodo National Park	0.3	2.7	0.2	14.3
124386	Lake Baikal	1332.6	1044.7	4.8	10.9
10904	Lake Malawi National Park	0.0	1.3	0.0	2.6
145586	Lake Turkana National Parks	0.0	0.0	0.3	0.0
124388	Laponian Area	1.1	15.0	0.1	1.3
198300	Laurisilva of Madeira	0.4	2.5	1.3	3.1
198298	Lorentz National Park	19.2	43.3	1.7	0.6
2570	Los Glaciares National Park	2.8	2.1	0.5	0.8
61610	Los Kat¡os National Park	1.6	11.8	0.2	1.4
900006	Maloti-Drakensberg Park	2.7	45.3	0.9	8.1
2577	Mammoth Cave National Park	2.4	10.4	1.2	2.2
10907	Mana Pools National Park, Sapi and Chewore Safari Areas	2.6	3.0	0.3	0.5
10745	Manas Wildlife Sanctuary	4.3	2.1	2.6	0.4
16792	Manovo-Gounda St Floris National Park	2.6	5.1	0.1	0.3
17760	Manu National Park	45.9	30.7	0.3	0.5
145583	Morne Trois Pitons National Park	0.4	2.7	0.8	0.9
20399	Mosi-oa-Tunya / Victoria Falls	0.0	0.2	0.0	0.2

18863	Mount Athos	13.1	0.7	5.8	6.1
124384	Mount Emei Scenic Area, including Leshan Giant Buddha Scenic Area	0.8	5.0	0.4	0.9
26654	Mount Huangshan	0.1	9.5	0.1	1.7
145585	Mount Kenya National Park/Natural Forest	3.1	11.0	0.3	1.2
2574	Mount Nimba Strict Nature Reserve	1.5	21.7	1.1	3.8
17050	Mount Taishan	0.8	0.4	1.2	1.0
198295	Mount Wuyi	7.2	122.2	1.0	8.8
2005	Nahanni National Park	49.9	34.3	2.4	1.4
16793	Nanda Devi and Valley of Flowers National Parks	0.0	0.1	0.0	0.1
2015	Natural and Cultural Heritage of the Ohrid region	5.9	11.7	3.7	2.5
2010	Ngorongoro Conservation Area	17.2	0.3	1.6	0.3
2580	Niokolo-Koba National Park	19.2	10.5	1.7	1.7
220295	Noel Kempff Mercado National Park	57.9	235.9	0.4	5.4
124389	Okapi Wildlife Reserve	58.9	123.9	0.4	2.2
2579	Olympic National Park	21.5	249.2	0.7	6.4
220297	Pantanal Conservation Complex	4.6	6.6	0.5	0.4
198291	Peninsula Valdes	0.0	0.0	0.0	0.0
9613	Pirin National Park	0.4	10.4	0.2	1.6
2016	Plitvice Lakes National Park	0.8	3.6	0.3	0.6

198299	Puerto-Princesa Subterranean River National Park	0.3	7.1	0.6	1.7
145590	Pyrenees - Mont Perdu	0.0	0.9	0.1	0.2
26651	R¡o Abiseo National Park	12.7	16.5	0.7	1.0
5002	R¡o Platano Biosphere Reserve	365.6	252.0	8.5	10.1
4325	Redwood National and State Parks	4.2	94.4	0.8	6.8
61608	Rwenzori Mountains National Park	6.4	27.5	0.8	2.9
2007	Sagarmatha National Park	0.0	0.0	0.1	0.1
10906	Salonga National Park	65.9	96.8	0.2	0.7
9614	Sangay National Park	2.1	22.7	0.1	1.2
5005	Selous Game Reserve	127.2	77.8	0.8	1.6
2575	Serengeti National Park	0.4	4.1	0.0	0.6
67724	Shark Bay, Western Australia	5.8	2.7	12.4	14.3
68915	Shirakami-Sanchi	0.0	4.2	0.0	0.5
20062	Sian Ka'an	16.8	35.4	1.1	2.0
2006	Simien National Park	0.0	0.0	0.2	0.0
16791	Sinharaja Forest Reserve	0.2	5.3	0.2	0.8
9612	Srebarna Nature Reserve	0.0	0.4	0.3	1.2
902368	St Kilda	0.0	1.6	0.2	0.8
14177	Sundarbans National Park	0.0	0.0	0.0	0.0
5003	Tai National Park	1.6	104.0	0.1	5.3
10903	Talamanca Range-La Amistad Reserves / La Amistad National Park	11.4	146.0	0.2	2.3

5000	Tasmanian Wilderness	90.4	5.9	0.8	0.6
4999	Tassili n'Ajjer	0.0	0.0	0.0	0.0
26652	Te Wahipounamu South West New Zealand	41.1	46.0	0.3	3.0
145580	The Sundarbans	0.0	0.0	0.0	0.0
67729	Thungyai - Huai Kha Khaeng Wildlife Sanctuaries	10.6	32.3	0.2	1.2
197	Tikal National Park	0.3	77.2	0.1	6.6
26649	Tongariro National Park	0.3	103.9	0.1	10.2
26653	Tsingy de Bemaraha Strict Nature Reserve	0.8	5.1	0.1	0.8
67726	Ujung Kulon National Park	1.3	5.0	0.2	2.5
900010	Uluru-Kata Tjuta National Park	0.0	0.0	0.0	0.0
93294	Virgin Komi Forests	41.2	83.2	0.2	0.8
2017	Virunga National Park	110.1	237.3	3.1	5.3
124387	Volcanoes of Kamchatka	70.1	12.7	0.4	0.1
124385	W National Park of Niger	0.0	0.0	0.0	0.0
93295	Waterton Glacier International Peace Park	540.7	317.1	23.1	14.9
198301	Western Caucasus	3.9	10.8	0.2	0.4
17757	Wet Tropics of Queensland	72.6	101.4	0.9	2.8
68918	Whale Sanctuary of El Vizcaino	0.0	0.0	0.0	0.0
10902	Wood Buffalo National Park	2581.5	513.4	11.7	8.9

67731	Wulingyuan Scenic and Historic Interest Area	0.4	1.4	0.2	0.3
68916	Yakushima	0.0	1.1	0.0	0.3
2013	Yellowstone National Park	217.0	59.4	6.3	3.1
10908	Yosemite National Park	45.4	32.0	3.8	3.1

Chapter 5Supplementary Material for "Gaps and opportunities for the World Heritage Convention to contribute to global wilderness conservation"

Table S5.1. Coverage of global-scale wilderness in Mixed and Natural World Heritage Sites

WDPAID	*WHS Name*	*Area of WHS (km2)*	*Area Wilderness (km2)*	*% of WHS Wilderness*
555512003	Putorana Plateau*	19,757	19,800	100
2005	Nahanni National Park*	4,825	4,827	100
220298	Central Suriname Nature Reserve	16,236	16,029	98.7
900878	Purnululu National Park	2,443	2,343	95.9
10902	Wood Buffalo National Park	45,348	43,112	95.1
220297	Pantanal Conservation Complex	1,987	1,830	92.1
67727	Air and Tenere Natural Reserves	78,520	70,523	89.8

220293	Gunung Mulu National Park	526	472	89.7
902480	Dong Phayayen-Khao Yai Forest Complex	6,205	5,227	84.2
900629	Central Sikhote-Alin	3,990	3,339	83.7
555577555	Okavango Delta	20,505	17,105	83.4
5000	Tasmanian Wilderness World Heritage Area	15,829	12,716	80.3
220296	Central Amazon Conservation Complex	51,198	39,809	77.8
2018	Kluane / Wrangell-St Elias / Glacier Bay / Tatshenshini-Alsek	97,066	74,968	77.2
17760	Manu National Park	17,013	12,520	73.6
220295	Noel Kempff Mercado National Park	16,178	11,462	70.9
26653	Tsingy de Bemaraha Strict Nature Reserve	1,571	1,081	68.8
61612	Canaima National Park	28,954	19,388	67.0
17759	Gros Morne National Park	1,803	1,118	62.0
2580	Niokolo-Koba National Park	8,265	4,996	60.4
93294	Virgin Komi Forests	28,639	17,022	59.4
555547988	Sangha Trinational	7,510	4,017	53.5
555547991	Lena Pillars Nature Park	13,167	7,023	53.3
124388	Laponian Area	9,267	4,846	52.3

2570	Los Glaciares National Park	7,170	3,645	50.8
16792	Manovo-Gounda St Floris National Park	18,829	9,106	48.4
26689	Canadian Rocky Mountain Parks	23,529	11,270	47.9
26652	Te Wahipounamu South West New Zealand	25,083	11,256	44.9
67729	Thungyai - Huai Kha Khaeng Wildlife Sanctuaries	6,327	2,838	44.9
168241	Golden Mountains of Altai	17,226	7,418	43.1
18337	Hawaii Volcanoes National Park	846	364	43.0
900010	Uluru-Kata Tjuta National Park	1,331	549	41.2
555556049	Tajik National Park (Mountains of the Pamirs)	25,351	10,086	39.8
555556048	Namib Sand Sea	30,824	12,037	39.1
198298	Lorentz National Park	23,655	8,942	37.8
124387	Volcanoes of Kamchatka	39,738	14,828	37.3
2012	Everglades National Park	5,840	2,157	36.9
2572	Kakadu National Park	19,211	6,308	32.8
555547987	Lakes of Ounianga	631	195	30.9
124386	Lake Baikal	85,317	23,613	27.7
4999	Tassili n'Ajjer	75,543	17,615	23.3

220292	Kinabalu Park	769	173	22.5
12202	Gondwana Rainforests of Australia	3,698	811	21.9
20062	Sian Ka'an	5,299	1,155	21.8
902347	Cape Floral Region Protected Areas	11,021	2,313	21.0
555512005	China Danxia	836	99	11.8
20388	Banc d'Arguin National Park	11,981	1,083	9.0
902335	Tropical Rainforest Heritage of Sumatra	25,919	1,469	5.7
900880	Uvs Nuur Basin	12,505	337	2.7
5002	Rio Plaitano Biosphere Reserve	5,078	47	0.9
145586	Lake Turkana National Parks	1,542	4	0.3
555556046	El Pinacate y Gran Desierto de Altar Biosphere Reserve	7,121	16	0.2

*Notes: * LoW area slightly exceedes World Heritage Site area in these two sites due to the 1km2 resolution of the wilderness map. Where a site is 100% wilderness, some pixels may extend beyond the sites borders.*

Table S5.2. Biorealms where global-scale wilderness is not protected within Natural and Mixed World Heritage Sites, and nationally designated protected areas with the largest coverage of wilderness within these "gap" biorealms

Realm	*Biome*	*WDPAID*	*Protected Area Name*	*Country*	*Area of Wilderness protected*	*% of PA wilderness*
Afrotropic	Mangrove	303879	Pongara	GAB	524	54.3
Afrotropic	Mangrove	301603	Mangroves along Msimbazi	TZA	529	44.1
Afrotropic	Mangrove	308634	Ndongere	CMR	597	25.5
Afrotropic	Montane grasslands and savannas	13759	Bale	ETH	1,326	11.2
Afrotropic	Montane grasslands and savannas	779	Nyika	MWI	2,526	80.5
Afrotropic	Montane grasslands and savannas	348	National Park Quciama	AGO	5,028	58.1
Afrotropic	Temperate grasslands savannas and shrublands	na	No Coverage	na	na	na
Australasia	Mediterranean forests woodlands and scrub	126263	Dundas	AUS	1,457	18.5
Australasia	Mediterranean forests woodlands and scrub	314891	Mount Manning Range	AUS	1,493	80.0
Australasia	Mediterranean forests woodlands and scrub	126757	Nuytsland	AUS	1,767	29.0
Australasia	Temperate grasslands savannas and shrublands	555548828	Gilmore	AUS	72	9.1

Australasia	Temperate grasslands savannas and shrublands	1130	Pilliga	AUS	85	10.2
Australasia	Temperate grasslands savannas and shrublands	63058	Idalia	AUS	890	62.8
Australasia	Tropical and subtropical dry broadleaf forests	na	No Coverage	na	na	na
Indo-Malay	Deserts and xeric shrublands	6699	Mahal Kohistan	PAK	572	87.2
Indo-Malay	Deserts and xeric shrublands	834	Kirthar	PAK	797	26.7
Indo-Malay	Deserts and xeric shrublands	6684	Runn of Kutch	PAK	6,677	62.8
Indo-Malay	Flooded grasslands and savannas	1857	Wild Ass	IND	2,537	35.2
Indo-Malay	Flooded grasslands and savannas	19683	Kachchh Desert	IND	7,327	53.7
Indo-Malay	Mangrove	30000	Sembilang	IDN	1,187	44.4
Indo-Malay	Mangrove	1490	Tanjung Puting	IDN	3,509	84.7
Indo-Malay	Mangrove	317262	Sebangau	IDN	4,649	77.1
Indo-Malay	Temperate broadleaf and mixed forests	26159	Namdapha	IND	632	16.7
Indo-Malay	Temperate broadleaf and mixed forests	26156	Dibang Valley	IND	1,417	77.8
Indo-Malay	Temperate broadleaf and mixed forests	312904	Hukaung Valley (Extension)	MMR	8,852	49.3
Indo-Malay	Temperate coniferous forests	26147	Lado	IND	374	37.6

Indo-Malay	Temperate coniferous forests	26159	Namdapha	IND	632	16.7
Indo-Malay	Temperate coniferous forests	26156	Dibang Valley	IND	1,417	77.8
Indo-Malay	Tropical and subtropical coniferous forests	306383	Upper Agno River Basin	PHL	60	7.5
Indo-Malay	Tropical and subtropical coniferous forests	8897	Lingga Isaq	IDN	647	74.3
Indo-Malay	Tropical and subtropical grasslands savannas	303303	Bardia - Buffer Zone	NPL	35	6.3
Indo-Malay	Tropical and subtropical grasslands savannas	555569944	Banke	NPL	94	16.5
Indo-Malay	Tropical and subtropical grasslands savannas	1308	Bardia	NPL	160	17.8
Nearctic	Deserts and xeric shrublands	555586215	Grand Staircase-Escalante	USA	1,909	53.9
Nearctic	Deserts and xeric shrublands	366624	Cabeza Prieta	USA	3,057	96.4
Nearctic	Deserts and xeric shrublands	1111190	Death Valley	USA	9,235	73.5
Nearctic	Mediterranean forests woodlands and scrub	11115320	Ventana	USA	303	32.5
Nearctic	Mediterranean forests woodlands and scrub	11115318	Sespe	USA	666	75.9
Nearctic	Mediterranean forests woodlands and scrub	374270	San Rafael	USA	752	97.0
Nearctic	Temperate broadleaf and mixed forests	555567267	Reserve aquatique projetee de la Riviere-Dumoine	CAN	1,428	98.9
Nearctic	Temperate broadleaf and mixed forests	366301	Boundary Waters Canoe Area	USA	3,969	89.3

Nearctic	Temperate broadleaf and mixed forests	66395	Quetico Provincial Park (Wilderness Class)	CAN	4,543	96.3
Nearctic	Tropical and subtropical coniferous forests	306852	Tutuaca	MEX	1,264	28.9
Nearctic	Tropical and subtropical coniferous forests	107620	C.A.D.N.R. 043 Estado de Nayarit	MEX	5,194	22.2
Nearctic	Tropical and subtropical dry broadleaf forests	na	No Coverage	na	na	na
Nearctic	Tropical and subtropical grasslands savannas	555512051	Lower Rio Grande Valley	USA	244	2.0
Nearctic	Tropical and subtropical grasslands savannas	12999	Sabine	USA	404	69.5
Neotropic	Deserts and xeric shrublands	9436	Pampa del Tamarugal	CHL	52	4.1
Neotropic	Deserts and xeric shrublands	478543	Area De Protecao Ambiental Lago De Sobradinho	BRA	207	1.7
Neotropic	Deserts and xeric shrublands	198366	Parque Nacional Serra Das Confusoes	BRA	3,682	44.4
Neotropic	Mediterranean forests woodlands and scrub	na	No Coverage	na	na	na
Neotropic	Montane grasslands and savannas	16845	Laguna Brava	ARG	404	9.6
Neotropic	Montane grasslands and savannas	555587116	De la Vicuna	ARG	3,224	11.6
Neotropic	Montane grasslands and savannas	36	Eduardo Avaroa	BOL	3,968	58.0
Oceania	Tropical and subtropical moist broadleaf forests	999916	Hawaii Volcanoes	USA	375	38.9
Palearctic	Flooded grasslands and savannas	14900	Siwa	EGY	4,057	52.5

Palearctic	Flooded grasslands and savannas	555543027	El-Qatara Depression	EGY	10,790	48.3
Palearctic	Mediterranean forests woodlands and scrub	555570044	Oued Chbeyka	MAR	843	63.6
Palearctic	Mediterranean forests woodlands and scrub	555544265	Msseyed	MAR	1,381	67.8
Palearctic	Mediterranean forests woodlands and scrub	555570012	Reserve a Outarde	MAR	2,154	8.7